

University of Alberta

Community-Level Response of Birds to Burned and Salvage-logged Forests

by

Curtis Andrew Stambaugh



A thesis submitted to the Faculty of Graduate Studies and Research in partial fulfillment
of the requirements for the degree of Master of Science

in

Wildlife Ecology and Management

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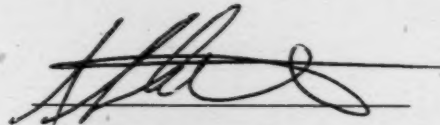
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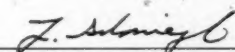
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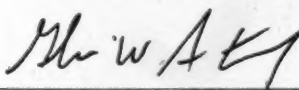
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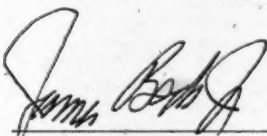
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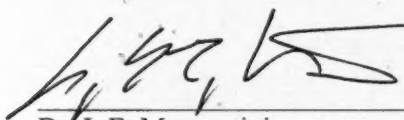
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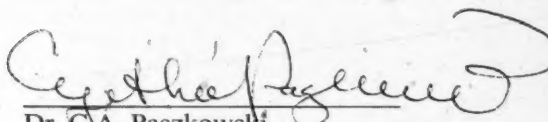
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ABSTRACT

Wildfire is the largest natural disturbance agent within the boreal forest. Understanding how wildlife respond to natural disturbances is important for conservation and to forest companies committed to sustainable forest management. This study monitored the forest bird community the first 3-years (1999-2001) following a wildfire, near Chip Lake, in west-central Alberta. The design included study sites within burned/unharvested (Leaves), burned/salvage-logged (Salvages), and unburned (Controls), deciduous-dominated mixedwood forest. By the second and third year post-fire, measures of songbird abundance and species richness were significantly higher within Leave treatments. Measures of relative reproductive activity indicated no difference or significantly higher indices for several species within burned forests. Similarity indices identified moderate overlap in bird communities between Leave and Control treatments, and suggested increased similarity over time; whereas divergence in similarity was indicated for bird communities in Salvage and Control sites. Many cavity-nesting species used natural cavities created by the fire, however the condition of the forest before disturbance may be very important, as many individuals used broken-top snags and/or snags in an advanced state of decay. This study suggests older-aged, burned forests provide a unique and valuable habitat for many breeding bird species, the quality of which is significantly diminished through salvage-logging. Forest management should recognize the importance of early post-fire habitat by maintaining unsalvaged areas throughout the landscape, as part of a coarse-filter approach to biodiversity conservation.

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Chapter 1: Research Rationale and Thesis Introduction

1.1 Background to Local and Regional Issues

Western Canada's boreal forest is sculpted in structure and composition by the size and frequency of various disturbance agents. Wildfire is the predominant natural disturbance in these forests, and has been for millennia (Gruell 1983; Agee 1993; Caton 1996). As a result, the boreal forest has sometimes been regarded as "fire-driven". Disturbance caused by wind, floods, and insect outbreaks (Kelsall et al. 1977; Bonan and Shugart 1989) also contribute to the heterogeneity of this system.

Variability in habitat types within the boreal mosaic allows for some of the richest breeding bird communities in North America (Robbins et al. 1986; Schieck et al. 1995; Kirk et al. 1996; Niemi et al. 1998; Hobson and Bayne 2000). The majority of these species are short-distance and Neotropical migrants (Robbins et al. 1989; Smith 1993), some from as far as South America (Thomas 1994). Birds are also the most abundant vertebrate taxon within the boreal forest (Smith 1993; Monkkonen and Viro 1997; Schmiegelow and Monkkonen 2002), and as a group, are increasingly being used as indicators for monitoring environmental change in a variety of systems (e.g., Diamond and Fillion 1987; Furness and Greenwood 1993).

Population trends over recent years have indicated significant declines in some Neotropical species, primarily due to anthropogenic disturbance (Robbins et al. 1989; Morton 1992; Thomas 1994; Rappole and McDonald 1994). Within the boreal region, forest harvesting poses the most immediate threat to wildlife (Imbeau et al. 2001), with resident birds potentially being some of the most sensitive species to these activities (Schmiegelow et al. 1997; Schmiegelow and Monkkonen 2002).

Currently, within the Province of Alberta, Canada, almost all public forested lands have been allocated for management under Forest Management Agreements (FMA's) and other forms of tenure. Advanced fire suppression technology and government policies, implemented to protect human infrastructure and future forest growth, strive to limit the dominant landscape influence of fire in many regions (Loope 1991; Niemela 1999). To a

large extent, the general public still perceive forest fires as destructive (Eberhart and Woodard 1987), and from a purely economic perspective, fire is simply considered wasteful. Increasing demand for timber resources has transformed logging into one of the most extensive disturbance agents affecting many parts of the boreal region (Schulte and Niemi 1998; Drapeau et al. 2000). Although this transition is most apparent throughout Fennoscandian boreal forests, where fire has been virtually eliminated, many Canadian jurisdictions are embarking on a similar management trajectory.

However, the ecological role of wildfire has not gone entirely unrecognized. Progressive forest harvesting strategies have adopted a coarse-filter approach to forest management, which attempts to emulate natural disturbance (Hunter 1993; Johnson et al. 1998; Hobson and Schieck 1999). This paradigm shift focuses on managing forests at the landscape-level, and attempting to simulate the pattern of fire, by maintaining stand ages within the natural range of variability, as approximated from historic fire rates. However, the long-term implications for wildlife, in assuming that logging can act as a surrogate for fire, are increasingly being questioned (e.g., Imbeau et al. 1999).

Although fires still burn large areas of forest in Alberta, in spite of significant investments in fire suppression, most commercially viable forest burned by fire are subject to harvest through timber salvaging operations. Where feasible to do so, salvage-logging occurs immediately after the fire is extinguished. It has become standard, once the perimeter of the fire is mapped, to remove (salvage) all merchantable timber left standing (burned and unburned) within the fire perimeter. Even though some snags and "fire-skips" (unburned patches) may be retained, salvage operations are usually much less progressive than the harvesting standard adopted by many companies that retain variable amounts of structure within cutblocks. In addition, such timber harvest is not necessarily included in Annual Allowable Cut (AAC) calculations, or subject to normal regeneration standards, and thus represents an economic boon to some companies.

Burned forests represent a unique early successional habitat type, characterized primarily by large volumes of standing dead trees (snags), increased vertical heterogeneity, and a flourish of new understory vegetation comprised primarily of pioneer species, that also increases horizontal heterogeneity. The implications of removing large quantities of this habitat from forested landscapes are a growing

conservation concern, but one that lacks much empirical research. Some wildlife species clearly utilize burned forest environments and some species being regarded as post-fire specialists, such as *Picoides* woodpeckers and certain bark- (*Scolytidae*) and wood-boring beetles (*Cerambycidae*) (Hutto 1995). Wood-boring beetles, in turn, impose economic losses on the salvaging of fire-burned timber, creating some ecological-economic conflict.

To date, woodpeckers and other cavity-nesting bird species have been the primary avian subjects on post-fire research, as many of these species appear to be dependent on attributes of burned forest (Hutto 1995; Hitchcox 1996; Caton 1996). Recently, some work has focused on the larger bird community within burned forests (see Hutto 1995; Imbeau et al. 1999; Hobson and Schieck 1999), but no study has monitored bird community succession, across habitat types, immediately following a wildfire within the boreal forest.

1.2 Thesis Overview

I report on the first 3 years of monitoring of the bird community, within an experimental framework, starting the first breeding season following a wildfire; in west-central Alberta's boreal mixedwood forest. Chapter 2 focuses on songbird assemblages, and compares burned and salvage-logged areas to unburned forest. This chapter also attempts to identify potential mechanisms behind songbird responses by constructing multivariate models that include a measure of food availability and various habitat characteristics. Chapter 3 involves an assessment of habitat quality by comparing species' relative reproductive activity among the 3 different habitat treatments (unburned/unharvested, burned/unharvested, and burned/harvested). Chapter 4 focuses specifically on the cavity-nesting bird community, by examining nest-cavity succession and turnover, as well as nest selection characteristics at 2 different spatial scales. Finally, chapter 5 summarizes the research findings, and suggests implications for management and directions for future research.

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Chapter 2: Comparison of Songbird Community Composition and Structure Among Burned, Unburned and Salvage-logged Areas

2.1 Introduction

Maintaining the ecological integrity of the boreal region should mean ensuring natural processes are maintained, as well as the wildlife populations they support. Current management philosophies, under the natural disturbance paradigm (Alberta Environmental Protection 1999), stress the importance of maintaining landscape attributes across a natural range of variability. Inherent to this philosophy is the hypothesis that through maintenance of a range of habitat features, wildlife and the ecological function of the system would also be maintained.

Stand replacement fires in the boreal forest create important structural legacies that should be considered features (habitats) to be maintained under the current paradigm. Structural legacies of fire, including live and dead woody material, create habitats much different from human-disturbed (harvested) areas (Hansen et al. 1991). In Alberta, Hobson and Schieck (1999) studied differences in bird communities between early-successional cutblocks and early-successional post-fire forests, at 3 different intervals post-disturbance, concluding that post-fire habitats are unique. Other recent studies conducted in conifer forests of the Rocky Mountains (Hutto 1995), aspen (*Populus* spp.) forests in Minnesota (Schulte and Niemi 1998), and black spruce (*Picea mariana*) stands in eastern Canada (Imbeau et al. 1999) report similar results; early-successional communities after forest harvesting are different in composition from post-fire bird communities, most notably by the absence of many cavity-nesting species.

There is growing concern among ecologists that burned forests do not receive the management attention necessary to maintain the wildlife species dependent upon them. This concern arises from current fire suppression policies, but more importantly, the immediate and large-scale timber salvaging that follows most wildfires. Salvage-logging of burned areas typically focuses on mature and older-aged forests, leaving behind large portions of young forest of no commercial value. However, the mature and old-growth

components of burned forests are the most valuable to birds that require larger diameter trees and/or snags for nesting and foraging (Caton 1996). Furthermore, for species that specialize in burned forest environments, such as Black-backed (*Picoides arcticus*) and Three-toed (*P. tridactylus*) woodpeckers, the "mixedwood" component of boreal forests provide an effective complement of nesting and foraging opportunities in deciduous and conifer trees, respectively (Caton 1996; Murphy and Lehnhausen 1998; Hoyt 2000; D. Stepnisky, unpubl. data).

Increasing consumer demand and new technologies in the pulp and paper industry have made deciduous and mixedwood boreal forests an increasingly lucrative target for harvest (Cumming et al. 1994). As a result, it is important to better understand how bird communities are responding to the major disturbance agents (fire and industrial harvesting) within the boreal region. Increasing our knowledge of boreal systems is integral to bird conservation, given that mixedwood boreal forests contain the highest number of breeding bird species in Canada (Robbins et al. 1986; Schieck et al. 1995; Schieck et al. 2000).

While several studies have demonstrated the importance of early-successional post-fire forests to cavity-nesting species, this study reports on the response of the broader songbird community, 1-3 years (1999-2001) following a wildfire in west-central Alberta. Songbirds were surveyed in 3 treatment types: burned-unharvested, burned-salvage logged, and unburned-unharvested forest, all of which were located in older, deciduous-dominated mixedwood stands.

Here, I address the following research questions:

- (1) Does the relative abundance, species richness, or species similarity of breeding songbirds differ among burned, salvage-logged, and unburned treatments?
- (2) Does the relative abundance and species richness of breeding songbirds change overtime within each treatment, from 1 to 3 years following wildfire?
- (3) What mechanisms might drive the response of the songbird community within the different treatments?

2.2 Methods

2.2.1 Study Area

This study was conducted in the foothills natural region (Alberta Sustainable Resource Development 1996) of west-central Alberta, within the Weyerhaeuser (Edson) Forest Management Agreement. During the spring of 1998, an estimated 14,000 ha of forested land, primarily black spruce (*Picea mariana*) and tamarack (*Larix laricina*) bottomlands, were burned by wildfire. Approximately 2,200 ha consisted of older deciduous-dominated upland forest, sub-dominated by white spruce (*Picea glauca*). This forest was deemed merchantable for harvest, and subsequently salvage-logged in the winter of 1998/1999. This region also has a prior human footprint due to historical high-grading of white spruce. Furthermore, fire suppression techniques (e.g., fire retardants) employed to contain the wildfire ultimately limited its extent.

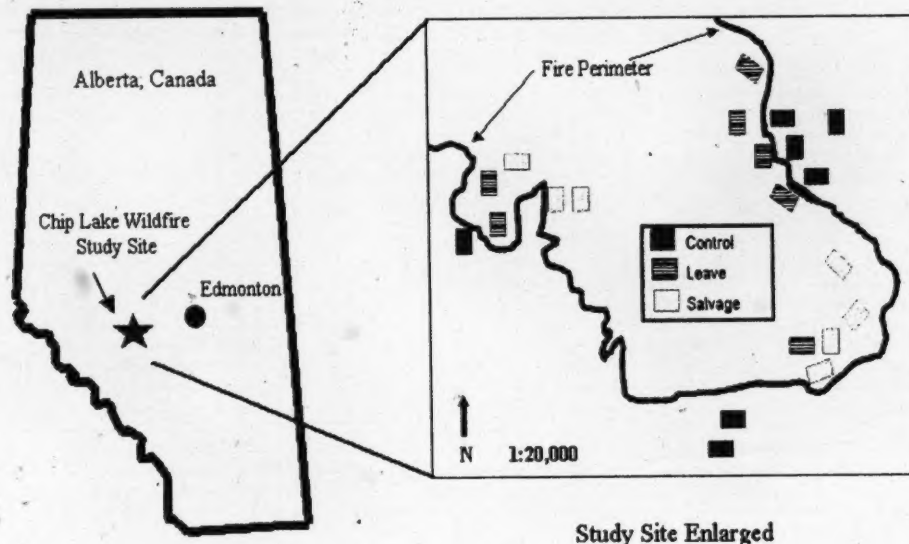
Twenty-one study sites, 10 ha in size (250 x 400 m), were selected across a 10 x 10 km township (Twp: 51, Rge: 09, W5M) (Figure 2-1). Seven sites were established as "Controls" (unburned and unharvested) and were located outside the burn perimeter in continuous forest. Seven sites were designated "Leaves" (burned and unharvested) and seven were designated "Salvages" (burned and harvested). All sites were selected from older aged (≥ 100 years) deciduous-dominated mixedwood stands, comprised primarily of trembling aspen (*Populus tremuloides*), balsam poplar (*Populus balsamifera*), and white spruce. Understory vegetation consisted of two dominant "tall shrub" species (obtaining heights ≥ 3 m): willow (*Salix* spp.) and green alder (*Alnus crispa*). Other dominant shrub species included bracted honeysuckle (*Lonicera invulcrata*), prickly rose (*Rosa acicularis*), currant and gooseberry, (*Ribes* spp.), and low-bush cranberry (*Viburnum edule*).

To the extent possible, treatment and control sites were spatially interspersed, separated by a minimum of 200 m. Sites were considered spatially independent for the use of inferential statistics (Hurlbert 1984).

2.2.2 Songbird Surveys

Measures of songbird abundance and composition were obtained from 100 m fixed-radius point counts. Two point count stations were located systematically within each of the 21 sites (Figure 2-2), for a total of 42 sampling stations. Observers passively documented all individual birds seen or heard within 50 m and 100 m radii, during a 5-minute interval. Each station was sampled multiple times throughout the breeding season, in each year: 7 visits/1999 (May 28 - July 7), 10 visits/2000 (May 27 - July 14), and 9 visits/2001 (May 27 - July 13). Surveys were conducted between sunrise and 10:00 am (Ralph et al. 1993) and only on days with little or no wind (~ 25 km/h or <5 on the Beaufort scale) and no rain. To minimize observer bias and diurnal variation, observers were rotated among study sites and survey start times varied throughout the morning period.

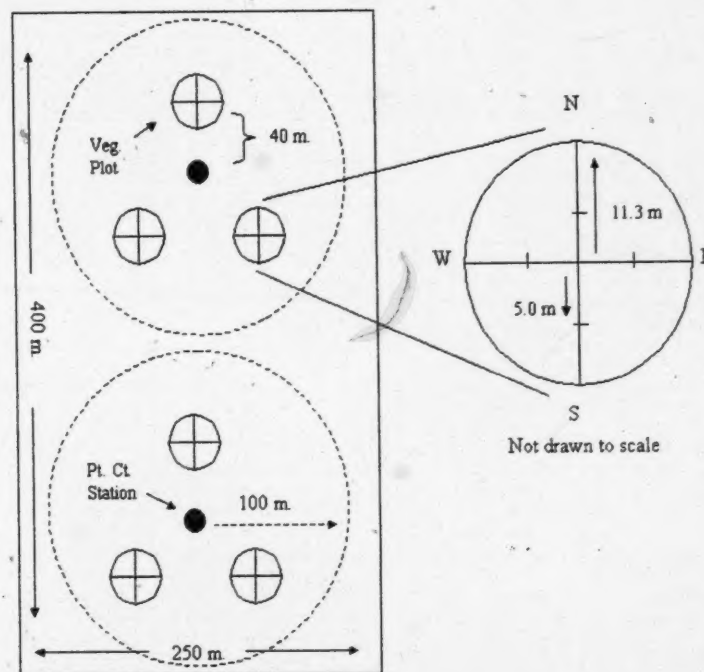
Figure 2-1: Location of the study site within Alberta, Canada and the experimental research design within the southeast corner of the 1998 Chip Lake wildfire.



2.2.3 Vegetation Sampling

Vegetative characteristics were collected from circular, 11.3 m radius plots (0.04 ha) systematically located within each 10 ha site. Plots were situated 40 m distance from the centre of each point count station, at 0, 120, and 240 degrees, for a total of 6 plots per site (Figure 2-2). Each plot was divided into four quadrants, marked at the 5 m radius point, denoting a 0.008 ha nested sub-plot. All measurements were taken in mid-July of each year.

Figure 2-2: Layout of the point count stations and vegetation sampling plots within each 10 ha site.



Within each plot, percent cover of grass and shrubs, and number of saplings (<2.5 cm diameter-at-breast-height – DBH) and poles (2.5-8 cm DBH) were recorded within the 0.008 ha sub-plot, noting the dominant species of tall and short shrubs; all other measures were recorded at the 0.04 ha scale. Tall shrubs were categorized as woody plants not found in the canopy, but capable of growing above breast height (1.3 m) (i.e. *Salix* and *Alnus* spp.). Short shrubs were categorized as woody plants found mostly

growing at or below breast height (1.3 m). All trees were recorded and placed into DBH categories, noting species, whether they were burned, showed evidence of wood/bark beetle presence, and whether cavities (natural and/or excavated) were present. Number of snags >12 cm DBH were recorded, including height, DBH, number and type of cavities, indication of burn and beetle evidence, and species (when possible). All snags <12 cm DBH were simply tallied. Coarse-woody debris (CWD) was recorded using a line-intersect method. One quadrant line, either N/S or E/W, was randomly selected and walked for its entire length (22.6 m), recording species (when possible), diameter at point of intersect, length, and rot class for all debris >1 cm diameter. Canopy and sub-canopy height was recorded using a clinometer and a burn intensity classification on a scale of 0-6, modified from Hutto (1995) (Appendix 1), was taken within each plot. Percent grass cover and CWD were added to the methodology in 2000, therefore, data from 1999 did not contain these metrics.

2.2.4 Invertebrate Collection

Food availability may be one of the mechanisms influencing avian abundance. Therefore, in 2001, terrestrial invertebrate sweep-net collections were taken systematically from 6 locations within each of the 21 sites, corresponding to the 0.008 ha vegetation sub-plots. Six samples per site were deemed adequate, as Song (1998) found that variation in biomass tended to asymptote after 5 samples, during invertebrate collections within mixedwood boreal forest. Collections were made using a 33 cm diameter sweep-net. Each sample consisted of 2 horizontal sweeps, approximately 3 m in length, at waist height, 5 m from the location origin, in 4 separate directions (NW, NE, SW, SE), for a total of 8 sweeps per sample. All insects collected were placed in specimen jars containing 75% ethanol for killing and preservation, and stored for lab analysis.

To help standardise environmental variability, sampling occurred only when ambient air temperature was between 15°C and 25°C, wind was <5 on the Beaufort scale, and vegetation was dry. All sampling was conducted after morning bird surveys, between 1000-1500 hours. Two sampling rounds were conducted at different stages of

the breeding season. Round 1 coincided with bird pairing and nesting (Jun/15 – Jun/21) and round 2 with nestling and fledging stages (Jul/06 – Jul/09).

Insects in all samples were counted and keyed to Order and Sub-order (some Sub-orders were keyed to Family) using Brues and Melander (1915), Essig (1954), and Borror et al. (1981). Order/Sub-order collections were then pooled for each site and oven-dried at 50°C for 24hrs, following Zach and Falls (1979) and Song (1998). Dry weight biomass was recorded to the nearest 0.001 gram per Order/Suborder per site.

2.2.5 Statistical Analysis

Prior to analyses, all data were tested for normality using Q-Q plots and for homogeneity of variance using box plots and the Levene statistic (Conover 1980; SPSS Inc. 1999). If necessary, data were transformed to satisfy assumptions for parametric statistical testing. If test assumptions could not be satisfied, non-parametric tests were used. Null hypotheses for among treatment effects were tested using 1-way Analysis of Variance (ANOVA), independently for 1999, 2000, and 2001. Significant results were adjusted for multiple, pairwise comparisons using the Bonferroni post-hoc procedure (Winer et al. 1991). Repeated Measures Analysis of Variance (RM ANOVA) was used to test null hypotheses for yearly differences within Controls, Leaves, and Salvages. Where a significant trend was evident, *t*-tests were used to identify specific changes from 1999-2000 and 1999-2001. Due to the proximity of point count stations within each 10 ha site, all "within-site" measures of the bird community were averaged to obtain a single "mean" value per site ($n = 21$), avoiding the possibility of double counting certain individuals.

Due to small sample sizes and low associated power, alpha (α) was set at 0.10 to reduce the chances of committing a Type II error. Arguably, a Type II error in applied research can be more costly for conservation efforts than Type I errors (Peterman 1990; Smith 1995; Schmiegelow et al. 1997; Johnson 1999). Statistical procedures were carried out using SPSS 10.0.3 for Windows (SPSS Inc. 1999). Power analyses for statistical tests were conducted using GPOWER (Faul and Erdfelder 1992).

Relative Abundance

Measures of relative abundance were derived from the 100 m radius point counts, focusing on passerine (songbird) species, and excluding all raptors, corvids, shorebirds, grouse, and woodpeckers. These groups were excluded due to the inherent weakness of the point count methodology in detecting wide-ranging species. In addition, all individuals seen or heard flying over points were also excluded, as these detections did not confirm use of the site. All documented occurrences of individuals were included in the analysis by weighting observations based on the individuals "reproductive" behaviour (Table 2-1; Schmiegelow et al. 1997). The weighted index was then summed for each point count station and divided by the number of visits.

Table 2-1: Weighted index assigned to all bird observations during point count surveys in 1999, 2000, and 2001 (from Schmiegelow et al. 1997).

Behavioural Activity	Weight
Family group or juvenile	2.0
Adult carrying food	2.0
Pair observed	2.0
Nest site observed	2.0
Singing or countersinging male	1.0
Territorial dispute	1.0
Adult calling or observed	0.5

I chose not to use conventional measures of relative abundance, which assume singing males are paired and/or calculate the maximum number of individuals observed during any one visit, for the following reasons: 1) the vast majority of observations obtained by point counts are singing males, 2) unpaired males have been observed to sing more frequently than paired males (Gibbs and Faaborg 1990; Gibbs and Wenny 1993), increasing their chances of detection, 3) marginal habitat can result in poor pairing success (Villard et al. 1993), and 4) I am mainly interested in treatment differences. Hence, conventional measures can potentially mask a true habitat response and reduce the chances of detecting a treatment effect (Type II error). The weighted index increases the confidence of detecting a treatment effect, by taking into consideration territorial, non-territorial, and paired individuals.

Species Richness

I derived 2 separate measures of species richness. The first measure is an absolute tally of "all species" documented per treatment type for each of the 3-years, within the 100 m point count radii. Due to an unequal proportion of sampling across years, only 7 visits/station/year were used to derive an unbiased measure of richness between treatments and among years. In 1999, all 7 visits to each point were used. In 2000, 3 visits were dropped from the actual 10 rounds conducted. The last visit was excluded because it fell outside the 1999 survey interval. Two additional rounds (3rd and 6th) were also excluded, such that the remaining visits corresponded most closely with those in 1999. In 2001, 2 rounds (4th and 9th) were excluded from the 9 conducted, following the same procedure applied in 2000.

The second measure of richness only includes species that were considered "probable breeders". Only species observations that occurred more than once at greater than 10 day intervals were tallied, according to the International Bird Census Committee (IBCC) (1970) or observations that exhibited breeding evidence (i.e. ranking of 2 – see Table 2-1). For consistency, probable breeders were also tallied using only the 7 rounds/year established for "all species".

Species Similarity

Similarity of the bird community, between treatments, was calculated using the Morisita-Horn index (Magurran 1988), which is not sensitive to sample size or diversity (Wolda 1981). All calculations were made using EstimateS 5 (Colwell 1997). Based on past research indicating similar bird communities show gradual convergence, over longer temporal scales, I anticipated treatment bird communities in this study would also increase in similarity. One-sided Repeated Measures ANOVA was used to test species similarity between Leaves-Controls, Leaves-Salvages, and Controls-Salvages, over the 3-year period. *T*-tests were conducted to detect significant differences between years.

Food Availability

I predicted invertebrate biomass was higher within Leave treatments; therefore 1-sided ANOVA's were used to test for a mean difference among treatments. Simple linear regression was used to test if songbird abundance was related to invertebrate biomass, as a measure of available food resources, therefore, treatment data were analysed collectively for measures obtained during 2001 (n=21).

Habitat Associations

Multiple linear regression was used to test whether songbirds responded to certain vegetation characteristics. I first reduced the number of independent variables by removing ones that displayed collinearity in a correlation matrix. Criteria for removal was based on correlation coefficients greater than ± 0.60 . Regression models were developed for each year of sampling (1999 to 2001), by pooling treatment data (n = 21). Backward elimination, including interaction terms, was used to determine the best-fit model. At each step, variables were excluded if $\alpha \geq 0.10$. A single regression model, examining songbird abundance as a function of burn intensity within Leave treatments, was also developed, as burn intensity was highly correlated with vegetation characteristics included in the multivariate models, and subsequently removed. However, because burn intensity is a variable of potential management significance, I was interested in exploring this relationship explicitly.

2.3 Results

2.3.1 Description of Community Composition

During the 3-years of the study, a total of 69 different bird species were documented from 42-point count stations, across all 3 treatment types (see Appendix 2). As expected, little change in species composition occurred within the Control treatments. A total of 31 different species were detected during the first year (1999) and 36 species were detected in both the second (2000) and third (2001) years. Of all the bird species present within the Controls, the 5 most abundant species in each year comprised 55%,

50%, and 49% of the entire community during the first, second, and third year, respectively.

Species composition within Leave treatments was slightly more dynamic. During the first year post-fire, 35 species were detected, whereas 45 and 44 species were recorded during the second and third year, respectively. Community evenness was very similar to the Control treatments with 54%, 49%, and 54% of the community comprised of the 5 most abundant species during the first, second, and third year, respectively.

Salvaged treatments supported the lowest species richness, with 25, 33, and 25 species detected during the first, second, and third year, respectively. It should also be noted that many of these species were only detected once or twice, with no evidence of breeding. This could be a function of sampling bias, as more visual detections are recorded in clearcut areas (e.g., of birds flying through). White-throated sparrow (*Zonotrichia albicollis*) and Lincoln's Sparrow (*Melospiza lincolni*) were the dominant species, comprising just under 50% of the entire community alone. The top 5 abundant species comprised 79%, 69%, and 79% of the community during the first, second, and third year post-disturbance, respectively.

Appendices 3 – 5 show the mean relative abundance and percentage of community composition for species recorded within each treatment, averaged over the 3 years of monitoring.

2.3.2 Relative Bird Abundance

In the first year post-fire (1999), no difference was detected in mean bird abundance among the 3 treatments (Table 2-2; Figure 2-3). By the second year post-fire (2000), relative bird abundance differed significantly among treatment types, with both Leaves and Controls supporting higher numbers than Salvages (Table 2-2; Figure 2-3). Mean relative abundance remained significantly different during the third year post-fire (2001); however, while Salvage treatments experienced a marked increase in numbers, almost matching abundances within the Controls, Leaves continued to support significantly greater numbers of birds than the Salvages.

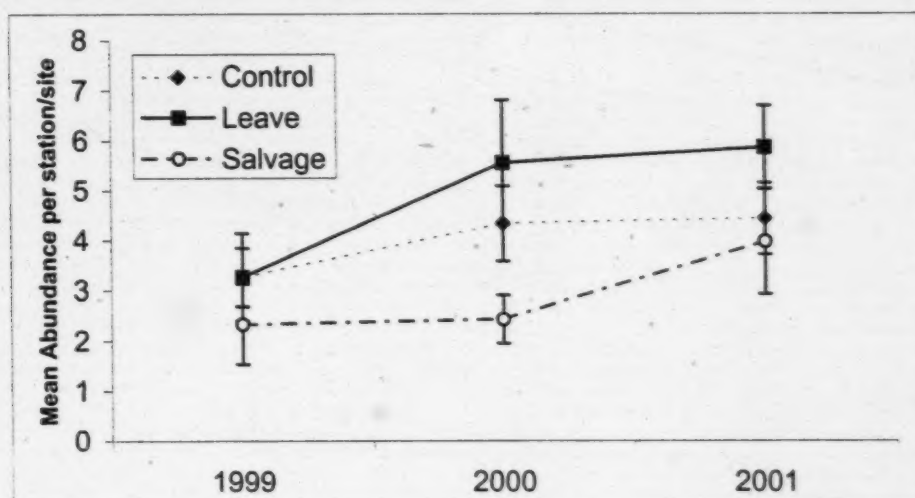
Table 2-2: Comparison of mean relative abundance (index) per 100 m sampling station/site between treatments, collected the first 3 years (1999-2001) after the Chip Lake wildfire (including $\pm 90\%$ CI). Bold type indicates significance at $\alpha \leq 0.10$.

Year	Treatments ¹			ANOVA		Bonferroni Post-Hoc	
	Control	Leave	Salvage	<i>F</i>	<i>p</i> ²	Multiple Comparison	<i>p</i> ²
1999	3.27 (0.58)	3.27 (0.87)	2.33 (0.80)	1.948	0.171	C = L	n/a
						C = S	n/a
						L = S	n/a
2000	4.34 (0.75)	5.56 (1.25)	2.43 (0.48)	13.493	< 0.001	C = L	0.295
						C > S	0.010
						L > S	< 0.001
2001	4.44 (0.72)	5.87 (0.83)	3.98 (1.01)	4.998	0.019	C = L	0.102
						C = S	1.000
						L > S	0.022

¹ Control = Unburned & Unharvested; Leave = Burned & Unharvested; Salvage = Burned & Harvested.

² Italics indicates value derived from square root transformed data.

Figure 2-3: Relative abundance among treatments and years ($\pm 90\%$ confidence intervals) following the Chip Lake wildfire. Control = unburned & unharvested; Leave = burned & unharvested; Salvage = burned & salvage-logged. 1999 = 1st Yr post-fire; 2000 = 2nd Yr post-fire; 2001 = 3rd Yr post-fire.



All treatments, including Controls, showed significant within-treatment increases in bird abundance from the first to third year post-fire and harvest (Table 2-3). Leaves and Controls displayed increases during the second year while Salvages didn't experience a significant increase until the third year (Table 2-3).

Table 2-3: Comparison of mean relative abundance (index) per 100 m sampling station/site among years within each treatment (including $\pm$ 90% CI), following the Chip Lake wildfire. Bold type indicates significance at $\alpha \leq 0.10$.

Treatment	Year ¹			RM_ANOVA		Pairwise Comparison		
	1999	2000	2001	F	p	Change	t ²	p ³
Control	3.27 (0.58)	4.34 (0.75)	4.44 (0.72)	8.463	0.005	99 < 00	-2.199	0.048
						00 = 01	-0.180	0.860
Leave	3.27 (0.87)	5.56 (1.25)	5.87 (0.83)	12.489	0.001	99 < 00	-2.922	0.013
						00 = 01	-0.409	0.690
Salvage	2.43 (0.80)	2.33 (0.48)	3.98 (1.01)	9.474	0.003	99 = 00	-0.221	0.829
						00 < 01	(-1.983)	0.047

¹ 1999 = 1st year Post-fire; 2000 = 2nd year Post-fire; 2001 = 3rd year Post-fire.

² Value in brackets indicates Mann-Whitney Z-statistic.

³ Value in italics derived from non-parametric test.

2.3.3 Species Richness

Leaves supported a significantly higher number of species than Salvages, in all 3 years post-fire and harvest, indicated by each measure of species richness (Table 2-4; Figure 2-4a-b). Leaves also supported significantly more species than Controls during the second and third year post-fire; however, the number of potentially breeding species did not differ from Controls. The difference in results between measures may indicate that more species utilize burned areas over unburned areas for reasons other than nesting (i.e. foraging).

Table 2-4: Comparison of the mean number of species per 100 m radius sampling station/site among treatments, collected the first 3 years (1999-2001) after the Chip Lake wildfire (including $\pm$ 90% CI). Bold type indicates significance at $\alpha \leq 0.10$.

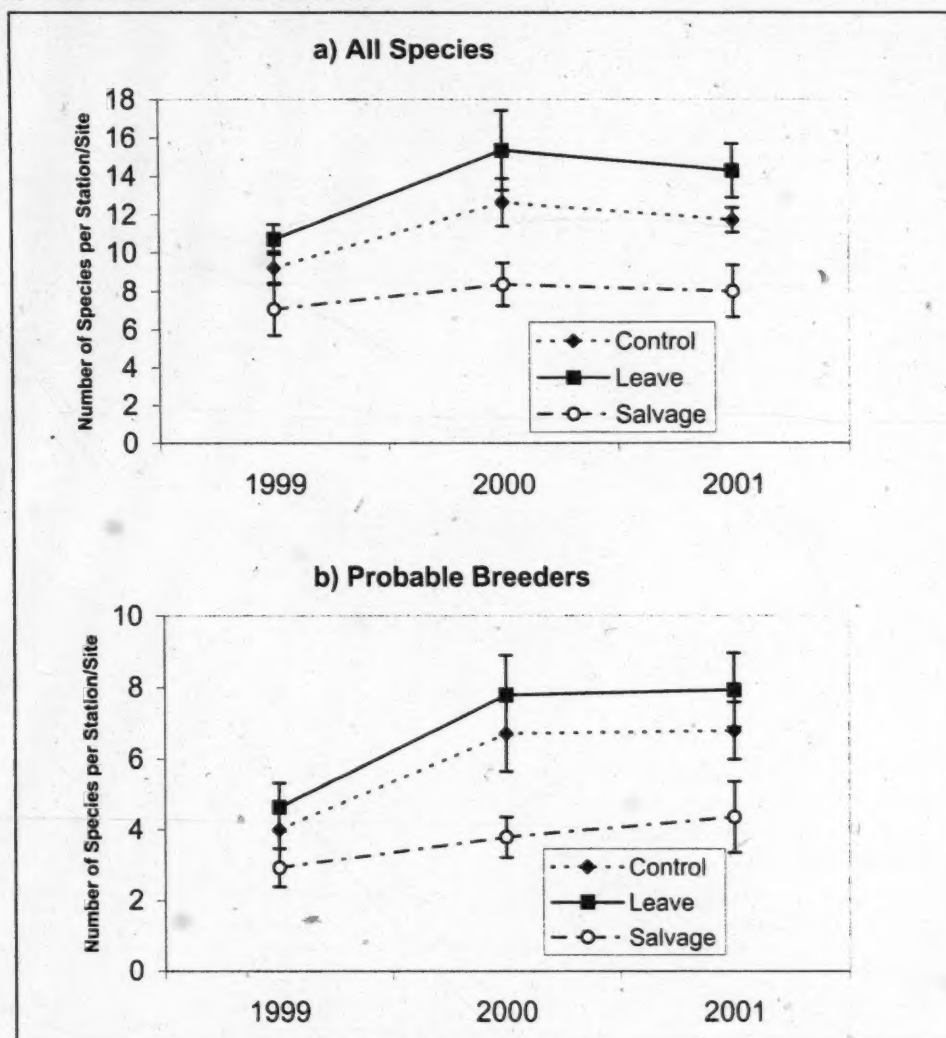
Year	Measure ¹	Treatments ²			ANOVA		Bonferroni Post-Hoc	
		Control	Leave	Salvage	F	p ³	Multiple Comparison	p ³
1999	All Spp.	9.2 (0.85)	10.7 (0.79)	7.1 (1.36)	9.832	< 0.001	C = L	0.231
							C > S	0.040
							L > S	< 0.001
	Breederers	4.0 (0.53)	4.6 (0.68)	2.9 (0.54)	6.809	0.003	C = L	0.536
							C > S	0.084
							L > S	0.002
2000	All Spp.	12.6 (1.24)	15.4 (2.08)	8.4 (1.11)	16.602	< 0.001	C < L	0.098
							C > S	0.004
							L > S	< 0.001
	Breederers	6.7 (1.05)	7.8 (1.11)	3.8 (0.57)	16.568	< 0.001	C = L	0.675
							C > S	< 0.001
							L > S	< 0.001
2001	All Spp.	11.7 (0.65)	14.3 (1.41)	8.0 (1.36)	22.012	< 0.001	C < L	0.031
							C > S	0.001
							L > S	< 0.001
	Breederers	6.8 (0.80)	7.93 (1.03)	4.4 (1.00)	11.756	< 0.001	C = L	0.410
							C > S	0.008
							L > S	< 0.001

¹ All Spp. = All species observations, excluding fly-overs; Breederers = Only species that showed evidence of territoriality or probable breeding activity.

² Controls = unburned & unharvested; Leaves = burned & unharvested; Salvages = burned & harvested.

³ p-values in italics are calculated using log transformed data.

Figure 2-4: Species richness among treatments and years ($\pm 90\%$ confidence intervals) following the Chip Lake wildfire. Control = unburned & unharvested; Leave = burned & unharvested; Salvage = burned & harvested. 1999 = 1st Yr post-fire; 2000 = 2nd Yr post-fire; 2001 = 3rd Yr post-fire.



Number of species within both Leaves and Controls experienced a significant increase from the first to second year post-fire, consistent for each measure of richness (Table 2-5). Salvages also experienced an increase in number of species among years; however, only the "probable breeders" measure of richness was significant between the first and second year.

Table 2-5: Comparison of the mean number of species per 100 m radius sampling station/site, among years for each treatment during the first 3 years after the Chip Lake wildfire (including $\pm$ 90% CI). Bold type indicates significance at $\alpha \leq 0.10$.

TRMT	Measure ¹	Year ²			RM_ANOVA		Pairwise Comparisons		
		1999	2000	2001	F	p	Change	t	p
Controls	All Spp.	9.2 (0.85)	12.6 (1.24)	11.7 (0.65)	13.634	< 0.001	99 < 00 00 = 01	-4.032 1.173	< 0.001 0.251
	Breeders	4.0 (0.53)	6.7 (1.05)	6.8 (0.80)	14.796	< 0.001	99 < 00 00 = 01	-4.072 -0.096	< 0.001 0.924
Leaves	All Spp.	10.7 (0.79)	15.4 (2.08)	14.3 (1.41)	9.332	0.001	99 < 00 00 = 01	-3.703 0.758	0.001 0.456
	Breeders	4.6 (0.68)	7.8 (1.11)	7.9 (1.03)	12.974	< 0.001	99 < 00 00 = 01	-4.292 -0.168	< 0.001 0.868
Salvages	All Spp.	7.1 (1.36)	8.4 (1.11)	8.0 (1.36)	1.736	0.205	99 = 00 00 = 01	n/a n/a	n/a n/a
	Breeders	2.9 (0.54)	3.8 (0.57)	4.4 (1.00)	7.238	0.023	99 < 00 00 = 01	-1.947 -0.887	0.062 0.383

¹ All Spp. = All species observations, excluding fly-overs; Breeders = Only species that showed evidence of territoriality or probable breeding activity.

² 1999 = 1st Yr Post-fire; 2000 = 2nd Yr Post-fire; 2001 = 3rd Yr Post-fire.

2.3.4 Species Similarity

Bird communities supported by each treatment type during the first 3-years following fire, and subsequent harvest, were quite different. In an overall comparison among treatments, the species assemblages of Leaves and Controls were most similar (Table 2-6). As predicted, Leaves and Controls increased in similarity from the first to third year post-fire, although the communities remained distinct. The remaining comparisons were either inconclusive or contrary to the hypothesis that all treatment types would increase in similarity, due to convergence over time (Table 2-6).

Table 2-6: Comparison of mean species similarity (including standard error) between treatments and among years following the Chip Lake wildfire. 1999 = 1st year post-fire; 2000 = 2nd year post-fire; 2001 = 3rd year post-fire.

Comparison	Morisita-Horn ¹			RM_ANOVA α (1)	Similarity 1 - 3 Yrs
	1999	2000	2001		
Leave (vs) Control	0.415 ^a (0.028)	0.459 ^{ab} (0.025)	0.481 ^b (0.026)	0.002	↑
Leave (vs) Salvage	0.426 ^a (0.024)	0.455 ^a (0.020)	0.357 ^b (0.023)	0.001	None
Control (vs) Salvage	0.241 ^a (0.019)	0.207 ^a (0.019)	0.150 ^b (0.015)	< 0.001	↓

¹ Values with different letters are significant ($\alpha = 0.05$)

2.3.5 Individual Species Abundance

Leaves experienced a significant increase in the number of Alder Flycatchers (*Empidonax alnorum*), Common Yellowthroats (*Geothlypis trichas*), and Tennessee Warblers (*Vermivora peregrina*) from the first to third year post-fire. Least Flycatcher (*Empidonax minimus*) increased over time, although no statistical difference was observed (Table 2-7). Red-breasted Nuthatch (*Sitta canadensis*) and Brown Creeper (*Certhis americana*) experienced significant increases from the first to third year post-fire; however, Controls also experienced a significant increase in numbers, reflecting a potential increase in the regional population (Table 2-7).

Salvages were the most dynamic treatment type, dominated by early-successional species, and displaying high species turnover. Common Yellowthroat, LeConte's Sparrow (*Ammodramus leconteii*), Clay-coloured Sparrow (*Spizella pallida*), and Lincoln's Sparrow all experienced a significant increase in numbers from the first to third year post-disturbance, while Dark-eyed Junco (*Junco hyemalis*) and White-throated Sparrow decreased in abundance (Table 2-7).

Table 2-7: Change in individual species abundance (index) per 100 m radius sampling station/site (including standard error), within treatments from the 1st (1999) to 3rd (2001) year following the Chip Lake wildfire. Leave = burned & unharvested; Salvage = burned & harvested; Control = unburned & unharvested.

Treatment	Species ¹	Year			ANOVA ² <i>p</i>
		1999	2000	2001	
Leave	Alder Flycatcher	0.001 (0.001)	0.027 (0.009)	0.031 (0.018)	0.091
Leave	Least Flycatcher	0.013 (0.009)	0.021 (0.012)	0.032 (0.014)	0.571
Leave	Common Yellowthroat	0.003 (0.002)	0.039 (0.016)	0.090 (0.031)	0.047
Leave	Tennessee Warbler	0.001 (0.001)	0.026 (0.015)	0.034 (0.017)	0.061
Leave	Red-breasted Nuthatch	0.011 (0.004)	0.039 (0.016)	0.064 (0.015)	0.028
Leave	Brown Creeper	-----	0.025 (0.009)	0.042 (0.015)	0.008
Salvage	Lincoln's Sparrow	0.075 (0.017)	0.109 (0.015)	0.202 (0.027)	0.004
Salvage	Le Conte's Sparrow	0.003 (0.002)	0.055 (0.020)	0.123 (0.025)	0.001
Salvage	Clay-coloured Sparrow	0.003 (0.003)	0.006 (0.003)	0.042 (0.018)	0.051
Salvage	Common Yellowthroat	-----	0.019 (0.012)	0.034 (0.015)	0.015
Salvage	Dark-eyed Junco	0.053 (0.010)	0.025 (0.007)	0.007 (0.003)	0.005
Salvage	White-throated Sparrow	0.087 (0.025)	0.072 (0.025)	0.066 (0.025)	0.647
Control	Red-breasted Nuthatch	0.013 (0.006)	0.054 (0.016)	0.077 (0.016)	0.006
Control	Brown Creeper	-----	0.007 (0.005)	0.015 (0.009)	0.168

¹ See Appendix 2 for scientific names

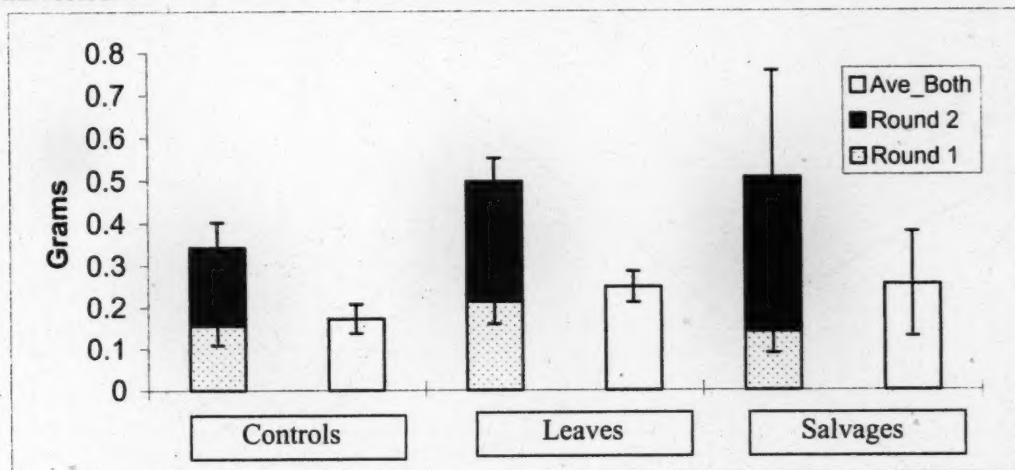
² *p* values derived from Kruskal-Wallis ANOVA

2.3.6 Food Availability

Leaves supported the greatest invertebrate biomass during the first round of collections; significantly higher than Salvages ($p = 0.098$), but no different from Controls ($p = 0.214$). During the second round of collections, the invertebrate biomass in Salvage treatments tripled (Figure 2-5), a result caused almost entirely by the emergence of

"Leafhoppers" (Suborder *Auchenorrhyncha*). Marked leafhopper increases were isolated to 2 Salvage sites, creating large within-treatment variation. Interestingly, these sites were the only 2 sites not prepared (mounded) for tree planting, and they boasted the greatest density of regenerating trembling aspen. Leaves, during the second round, experienced a slight increase over Controls ($p = 0.020$) but no longer differed from Salvages ($p = 0.274$). As an approximation of the food resources needed throughout the duration of a bird's breeding season, Leaves supported a significantly higher "average" biomass than Controls (1-sided $t = -2.672$, $p = 0.007$) but did not differ from Salvages (1-sided $t = -0.074$, $p = 0.471$). Appendix 6 contains the mean biomass for terrestrial invertebrates, grouped by Order, within each treatment.

Figure 2-5: Mean invertebrate biomass, to the nearest 0.001 gram ($\pm$ 90% CI), collected per site for each treatment during the 3rd year (2001) following the Chip Lake wildfire. Round 1 collected June 15-21/2001; Round 2 collected July 6-9/2001. Controls = unburned & unharvested; Leaves = burned & unharvested; Salvages = burned & harvested.



Linear regression identified a significant relationship between total invertebrate biomass and songbird abundance (Table 2-8). Variation within Salvage sites, due to leafhopper emergence, caused one outlier in the model. I felt it was not appropriate to remove this site, as I was interested in the relationship between invertebrate biomass and songbird abundance. However, because there is little information suggesting songbirds

effectively utilize leafhoppers due to the cost-benefits of small prey, especially to feed young (Burger et al. 1999), I repeated the analyses excluding all "leafhoppers" (Suborder *Auchenorrhyncha*). This resulted in a highly significant relationship, with substantially more variance explained (Table 2-8).

Table 2-8: Linear regression between songbird abundance (mean "index" per 100 m sampling station/site) and invertebrate biomass (mean "grams" per 6 collection plots/site) during the 3rd year (2001) following the Chip Lake wildfire. Bold type indicates significance at $\alpha \leq 0.10$.

Independent Variable	Intercept	Slope	R ²	p	95% CI of Slope	
					Min.	Max.
Total Invert. Biomass	3.767	2.216	0.150	0.083	-0.319	4.752
Biomass (excluding Leafhoppers)	2.125	8.252	0.377	0.003	3.156	13.349

2.3.7 Species-Habitat Associations

Songbirds responded significantly to coarse-woody debris (CWD), snags, % grass cover, % tall shrub cover, saplings, and trees between 23-38 cm DBH, depending whether it was the first, second, or third year post-fire and harvest (Table 2-9).

Invertebrate biomass, included in the model during 2001, was also identified as a significant variable, although it had a significant interaction with % grass cover and trees between 23-38 cm DBH. The interaction terms, % tall shrubs*snags in 1999 and saplings*trees 23-38 cm DBH in 2000, were also significant variables in the final models. Variation explained by each model was high (>65%) (Table 2-9). Songbird abundance also displayed a significant positive relationship with burn intensity, although variation explained in the model was relatively low (Figure 2-6). Appendices 7-9 provide a summary of different vegetative measures obtained from each treatment.

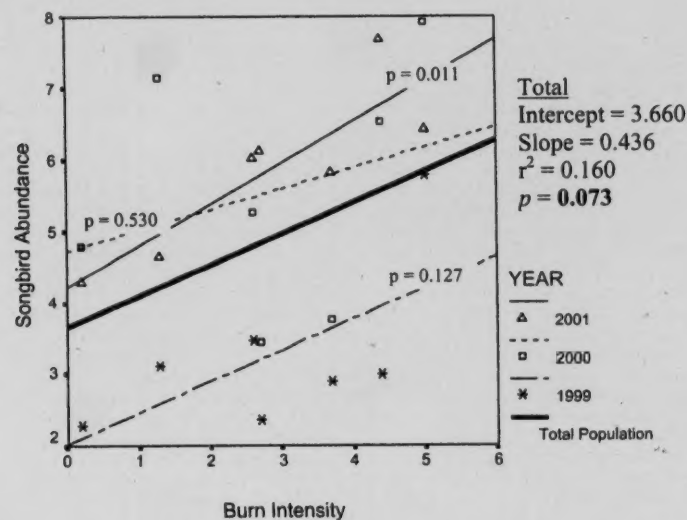
Table 2-9: Multivariate regression models, based on backward elimination, showing vegetation characteristics (including interactions) associated with songbird abundance (n=21) following the Chip Lake wildfire. Models were developed annually for the 1st (1999) to 3rd (2001) year post-fire. Bold type indicates significance at ($\alpha = 0.10$).

Variables	Regression ¹		
	1999	2000	2001
	($r^2 = 0.659$) ($p = <0.001$)	($r^2 = 0.770$) ($p = <0.001$)	($r^2 = 0.787$) ($p = <0.001$)
Intercept	1.903	0.040	2.560
Terrestrial Invert. Biomass	n/a	n/a	-3.163 (0.057)
CWD	n/a	3.868 E04 (0.001)	-----
Snags	ns	5.197 E05 (0.021)	4.902 E05 (0.001)
% Grass Cov.	n/a	0.061 (0.004)	ns
% Tall Shrub Cov.	ns	ns	0.062 (0.017)
Saplings (0-2.5cm DBH)	7.912 E06 (0.048)	ns	-----
Poles (2.5-8cm DBH)	ns	-----	-----
Trees 23-38cm DBH	ns	0.003 (0.002)	ns
Interaction Terms ²			
TallShrubs * Snags	1.255 E05 (<0.001)	-----	-----
Saplings * 23-38cm DBH	-----	-1.940 E07 (0.002)	-----
Invert. * %Grass Cov.	-----	-----	0.107 (0.003)
Invert. * 23-38cm DBH	-----	-----	0.003 (0.039)

¹ Dashed line indicates variables excluded from each model due to collinearity; n/a indicates data not available

² Only significant interactions are displayed

Figure 2-6: Relationship between songbird abundance (mean "index" per 100 m radius sampling station/site) and mean burn intensity per site (see Appendix 1 for burn classification) within Leave treatments for all 3 years (1999 – 2001) combined (n = 21) and individually for each year following the Chip Lake wildfire. 1999 = 1st Yr; 2000 = 2nd Yr; 2001 = 3rd Yr.



2.4 Discussion

Contrary to perceptions that wildfires reduce the wildlife habitat value of burned forests, this study suggests that post-fire forests have high habitat value for the forest bird community. Extensive songbird colonisation of post-fire forests (Leave treatments) was evident by the first full breeding season following disturbance. Although songbirds also colonised salvage-logged areas following harvest, comparisons between the two treatments revealed notable differences in community abundance and species richness.

2.4.1 Community Response to Burned and Salvage-Logged Areas

Older-aged, post-fire forests may be providing important resources to bird species, in order to support the higher songbird numbers observed in this study, at least during the first few year's post-fire. Relative songbird abundance within burned forest areas (Leaves) were as high as that supported by unburned areas (Controls) of similar size and composition, just 1 year post-fire. This pattern was mirrored in species richness,

while salvaged areas supported fewer individuals and less species, suggesting a high ecological value for older-aged post-fire forests. Although songbird abundance in both Leave and Control treatments increased over the 3-year study, perhaps reflecting regional population dynamics, Leaves experienced twice the increase over Controls during the second year post-fire, and maintained those levels throughout the third breeding season. While songbird numbers within Salvaged areas also increased substantially by the third year (no mean difference between Controls), they remained significantly lower than Leaves.

Results from this study corroborate some emergent patterns from research conducted on post-disturbance systems, even though site-specific conditions and harvesting practices may be inherently different. In Northeastern Minnesota, USA, Schulte and Niemi (1998) reported both bird abundance and species richness to be higher within burned forests 2-3 years post-fire, than in logged forests of the same origin. Hitchcox (1996) studied the post-fire response of cavity nesting birds within old conifer forests of the Rocky Mountains in Montana, USA, reporting that nesting densities increased with time post-fire. However, in black spruce dominated forest in Quebec, Imbeau et al. (1999) found species richness did not vary among different aged forests originating from fire, and bird abundance was highest in recent clearcuts; although, burned stands did comprise distinct bird communities, as clearcuts lacked the presence of the cavity-nesting guild. In coniferous forests of Washington, USA, Kreisel and Stein (1999) reported similar species numbers between burned and unburned areas during the non-breeding season, but again noted distinctly different bird assemblages between treatments.

Similarity indices also indicate different bird communities among burned, unburned, and salvage-logged areas in this system. These results are similar to Hobson and Schieck (1999), who noted the strongest distinction between post-fire and post-harvest bird communities in boreal mixedwood forest was immediately following disturbance, prior to gradual community convergence throughout the first 28 years. Despite immediate differences in community assemblage, this study also suggests an increase in species similarity between burned and unburned habitats even within the first 3 years post-fire. Salvaged areas did not exhibit the same pattern of convergence with

Controls. In fact, this study indicates divergence between community types from the first to third year following disturbance. This divergence over time may have resulted from a lag in species response to the harvested areas. Species returning from migration the first year after harvest, finding their old territory altered, may not have had sufficient time to locate replacement habitat, and may have settled within the salvage-logged areas. In subsequent years, individuals simply may not have returned to the area. Nevertheless, communities supported in the salvaged areas were distinct with respect to bird community assemblage, being dominated by 3 species comprised of one foraging and nesting guild (see Appendix 5). Salvage operations were not intended to emulate fire pattern through retention of structure within cutblocks, thus the resultant site types may be sufficiently distinct that bird communities may not converge.

This study indicates that quantifiable differences do occur at the community level between burned forests left standing and those that have been salvage-logged. It is also interesting that very similar patterns emerge from different systems undergoing stand-replacement fire even under very different circumstances. This and other recent studies tend to suggest commonalities in species response to post-fire forests, supporting the ecological value of such habitat to bird communities – habitat value that is significantly reduced by salvage-logging.

2.4.2 Species-Level Responses

Notable differences in species assemblages were evident between Leave and Salvaged areas. Leaves supported many more species typically associated with forest environments than did the salvaged areas, including Yellow-rumped Warbler (*Dendroica coronata*), Blue-headed Vireo (*Vireo solitarius*), Warbling Vireo (*V. gilvus*), Rose-breasted Grosbeak (*Pheucticus ludovicianus*), and Ovenbird (*Seiurus aurocapillus*) (see Appendix 4). Presence of these species is likely related to fire skips and areas of low intensity burn that result in maintenance of patches of relic forest for canopy dwelling species. Some Leave sites were also contiguous with unburned forest along the fire perimeter and some individuals may have established territories that overlapped burned and unburned habitat.

Leaves also supported a higher diversity of foraging and nesting guilds. Most notable was the presence of aerial foragers, such as the Western Wood-Pewee (*Contopus sordidulus*), Alder- (*Empidonax alnorum*), and Least Flycatcher (*E. minimus*). These species were observed using bare branches of burned trees as singing perches, while they hawked for insects. Alder Flycatcher, Common Yellowthroat, Tennessee Warbler, Red-breasted Nuthatch, and Brown Creeper all increased significantly over time in the Leaves; Nuthatches and Creepers did not increase exclusively within the Leaves, but their presence as members of the cavity-nesting guild enhanced the overall diversity within the burned forest habitats. These results are similar to Hutto (1995), who also reported burned forests were different in composition, especially due to the abundance of 3 foraging guilds: woodpeckers, flycatchers, and seedeaters.

Salvaged sites were dominated by species that nested and foraged on the ground and/or in shrubs and the communities were almost exclusively comprised of White-throated, Lincoln's, and Le Conte's Sparrows. Cavity-nesting House Wrens (*Troglodytes aedon*) were relatively common in the salvaged areas during the first year post-disturbance, using slash-piles left behind from the salvaging operations. However, their numbers declined significantly in the second year, when the piles were burned to reduce fuel load and increase the area of regeneration within cutblocks. Common Yellowthroat, Lincoln's, Le Conte's, and Clay-coloured Sparrows increased significantly with increasing time post-harvest, in contrast to decreases in the Dark-eyed Junco. These species, however, essentially represent a single nesting and foraging guild.

Cavity-nesting species (not exclusively songbirds) utilized standing burned trees (snags) in both salvage and Leave sites; however, differences in species numbers and representation were observed. Very few snags remained within the harvested areas, although Tree Swallows (*Tachycineta bicolor*) and Mountain Bluebirds (*Sialia currucoides*) occurred in low numbers and were observed exclusively within the salvaged areas. Northern Flicker (*Colaptes auratus*) and American Kestrel (*Falco sparverius*) were observed in both Leave and Salvages but were not sampled appropriately due to their large ranging territories. Black-capped Chickadee (*Parus atricapillus*), Red-breasted Nuthatch, and House Wren were also observed within both site types, although no evidence of breeding was found for nuthatches and chickadees within salvaged areas.

Boreal Chickadee (*Parus hudsonicus*), Brown Creeper, and Northern-hawk Owl (*Surnia ulula*) were found exclusively within Leave sites, although hawk owls are not exclusive cavity-nesters. These results raise concerns regarding resident species. Resident cavity-nesting species in this study appeared to prefer unsalvaged areas and/or were found exclusively within them (not including unburned areas), versus migrant cavity-nesting species that utilized both. These results differ from Imbeau et al. (1999), who noted logged stands were distinctly different from burned forest stands due to the absence of cavity-nesting birds, although he did find that migrant species contributed to higher abundances within clearcuts. Nevertheless, diversity of cavity-nesting species within salvaged areas may have been greater if snag retention had been increased (see Dickson et al. 1983).

Post-fire "specialists", such as Black-backed (*Picoides arcticus*) and Three-toed Woodpeckers (*P. tridactylus*) as well as Hairy (*P. villosus*), and Downy Woodpeckers (*P. pubescens*) were all observed within burned areas. However, their dynamics within this study system are the subject of a complementary investigation.

2.4.3 Vegetation and Food Response Mechanisms

Understanding the factors underlying species response has occupied ecologists for years. Although not entirely independent of each other, factors believed to influence bird diversity and abundance are often approached at 2 levels: (1) geographic (regional biogeography, landscape configuration, and metapopulation structure) and (2) local (habitat characteristics including food and nesting resources) (Willson and Comet 1996).

The majority of past work on bird communities has focused primarily at the local level, by attempting to explain bird response due to vegetation characteristics (MacArthur and MacArthur 1961; James 1971; James and Wamer 1982; Mills et al. 1991; Schieck et al. 2000; many others) and/or food availability (Hutto 1981; Guinan and Sealy 1987; Holmes and Schultz 1988; Burke and Nol 1998). While there have been several recent landscape-scale assessments of bird populations (e.g. McGarigal and McComb 1995; Donovan et al. 1997), and trends associated with different levels of habitat disturbance and configuration, these studies also tend to include or make reference to local scale

characteristics. Willson and Comet (1996) suggest that local ecological factors, specifically density of understory vegetation and litter invertebrates, are more important in determining bird diversity and abundance than regional factors. I focused my analyses on local-level factors.

I found aspects of the understory vegetation, particularly percentage (%) of grass and tall shrub (*Salix* and *Alnus* spp.) cover, number of saplings, and coarse-woody debris (CWD), to be significant variables, along with the biomass of terrestrial invertebrates, correlated with songbird abundance. Habitat-based models, developed independently for the first, second, and third year post-fire and harvest, explained relatively high (>65%) amounts of variation in songbird abundance. During the first year (1999), songbirds responded positively to sapling density, as well as to the interaction between % tall shrub cover and number of snags. This response is most likely due to the high density of pioneer plant species (i.e., suckering trembling aspen) present immediately after the wildfire. The interaction between shrubs and snags may also be an indication of a post-fire response to new understory vegetation. Number of snags increased within burned areas and suckering *Salix* species also flourished in these areas after parent trees were killed by fire. Older-growth forests that typically contain higher densities of snags also tend to support larger *Salix* and *Alnus* shrubs, which provide valuable nesting, foraging, and escape cover for a variety of bird species (Ehrlich et al. 1988).

During the second (2000) year, % grass cover and CWD data were collected, and these variables proved to be significant characteristics in the model, along with large mature trees (23-38 cm DBH). The response of forest songbirds to larger trees is no real surprise, although large trees by themselves were not a consistently significant variable throughout the 3 years of monitoring. The inclusion of invertebrate biomass in the third year (2001), to represent food resources, interacted significantly with trees 23-38 cm DBH and % grass cover, probably because these characteristics are important as terrestrial invertebrate habitat. CWD was strongly correlated with invertebrate biomass, and because the latter was considered more valuable to songbirds, CWD was removed from the 2001 model. However, for management purposes, a measure of CWD is more tractable.

Interestingly, number of snags was significant to bird abundance only during the second and third years post-fire (i.e. increasing time post-fire). This result matches an increase in cavity-nesting and bark gleaning bird species (see Table 2-7). Woodpeckers were not included in this analysis, however, species dependent on existing cavities (i.e. House Wrens, American Kestrels, etc), including species that can excavate cavities into weakened or decaying snags (i.e. Chickadees and Nuthatches) were included, and were present in higher numbers during the second and third year's post-fire.

There is little doubt among ecologists that food supply can affect birds in many ways, from individual starvation to the creation of population fluctuations (Newton 1998). I found terrestrial invertebrate biomass, considered a general food source for the songbird community, to be higher within burned areas (Leaves) during the third year post-fire. Bird abundance and richness measures were also higher within Leaves, and regression analysis indicated a significant relationship between invertebrate biomass and bird abundance. Whether increased invertebrate biomass within these burned forests will be sustained over time is unclear. It is possible that the songbirds were exhibiting a short-term response to insect "outbreak" within these early-successional burned forests, similar to irruptions associated with outbreaks of insects such as Spruce Budworm (*Choristoneura fumiferana*) (e.g., Crawford and Jennings 1989).

Regression analysis also identified songbird numbers to be significantly related to burn intensity. Songbirds appear to favour post-fire forests burned at higher intensity over areas experiencing low ground fires. This trend is corroborated by abundance analyses among treatments, where areas of low burn intensity more closely resembled unburned areas, and unburned areas tended to support lower bird numbers than Leaves, albeit non-significantly. This relationship suggests tangible management potential, considering fires subject to salvage operations are initially mapped to delineate the burn perimeter, and this usually includes a broad classification of burn intensity within the fire. While the burn classification I used was different than that generally applied, some modifications to the classification could help prioritise areas for wildlife habitat conservation, within salvage logging operations.

2.4.4 Study Limitations

Achieving the perfect experimental design in wildlife science is nearly impossible. This study met with some unavoidable constraints. Site selection was based on two main criteria: (1) reduce variability within bird communities caused by different habitat types and (2) focus on "old-growth" forest targeted by forest companies for harvest, which also contain the bird species most heavily impacted by industry disturbance. Further constraints in site selection were imposed by the underlying landscape characteristics. The Chip Lake fire contained more black spruce and conifer-dominated forest than upland deciduous-dominated forest, reducing the area available for study. Obtaining authorization from Provincial authorities to leave large unharvested tracts of merchantable burned forest was difficult. Therefore, the number of potential sites that fit the selection criteria of size (10 ha) and structure (≥ 100 year deciduous-dominated mixedwood) was limited to 7 replicates, and sites could not be randomly allocated to treatment. One could, in fact, argue that the effective sample size is actually one, considering all sites were within a single wildfire.

Leave sites were also unavoidably non-uniform in design, due to placement criteria and harvesting operations. Sites varied in their distance to the fire perimeter, and ranged from being isolated by harvested areas to partially contiguous with unharvested "fire-skips" or burned black spruce. Whether this caused additional within-treatment variability is unclear. However, with the exception of the isolates, the remainder of sites capture a wide range of natural variability in spatial context, inherent to wildfires.

Finally, even with an alpha set to 0.10 for ANOVA tests, power of rejecting a false null hypothesis from these tests ranged from 44.2% - 96.7%, depending on the variation in observed estimates. *A priori* power analyses indicated treatment replication in this study would have needed to be 17 (alpha set at 0.10) to obtain a power of 80% in detecting an effect size of 0.40, as conventionally defined (Cohen 1992). As previously discussed, obtaining that level of replication was logistically impossible and probably impractical for field application. However, conventionally defined effect size is arbitrary and may not always be biologically relevant (Steidl et al. 1997). Therefore, I recommend caution with interpretation of these analyses.

2.5 Conservation Implications

Results from this study indicate clear short-term differences between bird utilization of unsalvaged burned forests and those that have been salvage-logged. Immediate post-fire forests represent distinct habitats comprised of different bird assemblages. Overall, this study indicates that areas of older-aged forest burned at medium to high intensity are most beneficial to the songbird community. As a result, managers involved in planning salvage operations after wildfire should consider maintaining burned areas when designing harvest strategies.

Where conservation of forest dwelling bird species is a concern, older-aged burned forests represent a more similar habitat and support a more similar bird community to old-aged unburned forests than salvaged-logged areas, immediately post-fire. This study also indicates an immediate, albeit gradual, increase in bird community similarity between burned and unburned forest, while the opposite occurred for salvaged areas. Maintaining unharvested burned forests may support a more abundant and diverse bird community from early to mid-succession, while allowing quicker vegetative succession to a seral stage more representative of the original forest. Thus, areas left natural after fire may develop more rapidly into habitats for birds dependent on old-growth stands, especially if salvage-logged areas prolong displacement of these species. Furthermore, maintaining cavity-nesting bird habitat – in the form of snags and fire-killed trees – may better maintain the composition of the forest bird community (Brawn et al. 1982).

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Chapter 3: Relative Reproductive Activity: Use of Call-Playbacks to Assess Habitat Quality Among Habitat Types

3.1 Introduction

Abundance or density measures of a species, or species assemblage, can provide important information on the general utilization of a particular habitat. However, such information can also lead to inappropriate management applications if density estimates are false indicators of habitat quality. Van Horne (1983) suggested that density might be higher in low-quality habitats (reproductive sinks) versus high-quality habitats (reproductive sources), particularly in passerine bird species, due to inter-annual variation and/or social dominance. Local passerine densities fluctuate annually, reflecting short-term variability in food resources, predation, and abiotic environmental conditions, and not necessarily habitat quality. In territorial species, if juveniles are forced to disperse and/or unfit individuals cannot defend a territory within preferred habitat, a larger proportion of non-breeding individuals may concentrate in low-quality habitats. Therefore, surveys that cannot, or do not, discriminate breeding and nonbreeding individuals may fail to provide a representative assessment, in terms of habitat quality and associated persistence of populations.

Vickery et al. (1992a), using an index of reproductive success, found no clear correlation between high density and high reproductive success in 3 grassland sparrows. Savannah Sparrows (*Passerculus sandwichensis*), in fact, were significantly more successful at low densities. Purcell and Verner (1998) reported that nesting success of California Towhees (*Pipilo crissalis*), through direct nest monitoring, was also lower in high-density areas. Long-term population persistence for some bird species depends on the provision of appropriate recommendations regarding habitat conservation. Therefore, in a management context, abundance and density measures should be complemented with some additional measure (index) of productivity or relative breeding success.

Various methodologies have been employed to obtain information on avian reproductive success and productivity (see Mayfield 1975; du Feu and McMeeking 1991; Vickery et al. 1992b; Buford et al. 1996; Dale et al. 1997). However, the majority of

methods require extensive time and effort on the part of the observer, limiting the scope and scale of the assessment. Rangen et al. (2000) further noted that acquiring reproductive indices of birds in forested areas, through passive observation, is limited by habitat complexity. Recently, Gunn et al. (2000) proposed a response eliciting technique that uses Black-capped Chickadee (*Parus atricapillus*) mobbing call-playbacks. Success with this particular method allows an observer to obtain an index of relative reproductive success at the community-level, and to apply the technique over large study areas, particularly forested environments.

In this chapter, I create an index of relative reproductive activity through a combination of data collection techniques modified from Gunn et al. (2000) and observation ranking schemes modified from Vickery et al. (1992b) and Schmiegelow et al. (1997). First, I evaluate this methodology with respect to its ability to provide evidence of breeding bird activity. I predicted that playbacks (response elicited technique) would increase the number of observations of breeding activity over point counts (passive technique). Second, I selected 7 focal bird species, representing a subsample of the forest bird community, that were commonly found within 3 different habitat treatments: burned-unharvested (Leave), unburned-unharvested (Control), and burned-salvage logged (Salvage) forests. Using the reproductive index developed for the 7 bird species, I compare breeding activity among the treatments as an indication of habitat quality.

3.2 Methods

3.2.1 Study Area

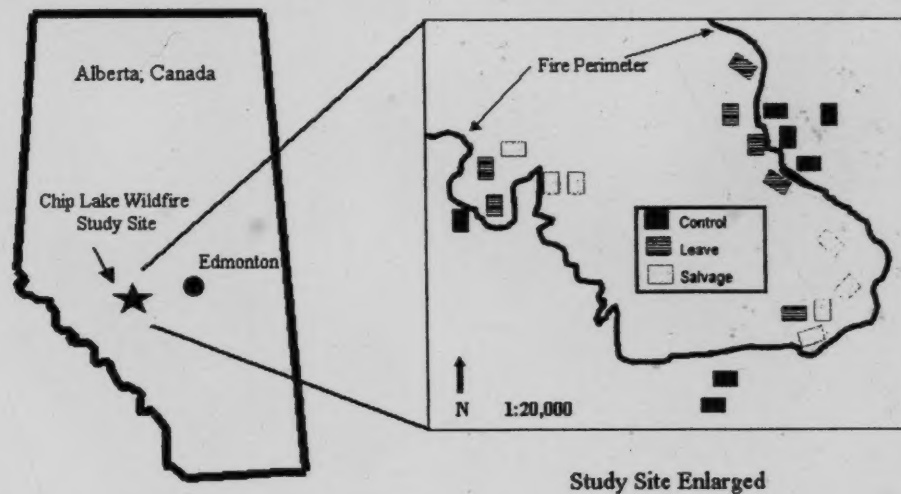
This study was conducted in the foothills natural region (Alberta Sustainable Resource Development 1996) of west-central Alberta, within the Weyerhaeuser (Edson) Forest Management Agreement. During the spring of 1998, an estimated 14,000 ha of forested land, primarily black spruce (*Picea mariana*) and tamarack (*Larix laricina*) bottomlands, were burned by wildfire. Approximately 2,200 ha consisted of older deciduous-dominated upland forest, sub-dominated by white spruce (*Picea glauca*). This forest was deemed merchantable for harvest, and subsequently salvage-logged in the

winter of 1998/1999. This region also has a prior human footprint due to historical high-grading of white spruce. Furthermore, fire suppression techniques (e.g., fire retardants) employed to contain the wildfire ultimately limited its extent.

Twenty-one study sites, 10 ha in size (250 x 400 m), were selected from a 10 x 10 km township (Twp: 51, Rge: 09, W5M) (Figure 3-1). Seven sites were established as "Controls" (unburned and unharvested) and were located outside the burn perimeter in continuous forest. Seven sites were designated "Leaves" (burned and unharvested) and seven were designated "Salvages" (burned and harvested). All sites were selected from older aged (≥ 100 years) deciduous-dominated mixedwood stands, comprised primarily of trembling aspen (*Populus tremuloides*), balsam poplar (*Populus balsamifera*), and white spruce. Understory vegetation consisted of two dominant "tall shrub" species (obtaining heights ≥ 3 m): willow (*Salix* spp.) and green alder (*Alnus crispa*). Other dominant shrub species included bracted honeysuckle (*Lonicera invulcrata*), prickly rose (*Rosa acicularis*), currant and gooseberry, (*Ribes* spp.), and low-bush cranberry (*Viburnum edule*).

To the extent possible, treatment and control sites were spatially interspersed, separated by a minimum of 200 m. Sites were considered spatially independent for the use of inferential statistics (Hurlbert 1984).

Figure 3-1: Location of the study site within Alberta, Canada and the experimental research design within the southeast corner of the 1998 Chip Lake wildfire.



3.2.2 Assessment of Reproductive Activity

I used Black-capped Chickadee (*Parus atricapillus*) mobbing call-playbacks in conjunction with 100 m fixed-radius point counts, modified from Gunn et al. (2000), to obtain evidence of breeding bird activity. Playbacks mimicked the auditory response of mobbing chickadees in the presence of a potential predator. Chickadee mobbing tends to elicit intra- and interspecific responses from many other forest birds, either to assist or observe in the mobbing (Hurd 1996). There are various theories to why certain species will mob a predator (see Curio 1978), though few have been tested empirically, and no insights are provided here.

Unlike point counts alone, which are primarily auditory surveys of singing males, playbacks typically provide an observer with increased visual observations. The observer then has the opportunity to document various breeding behaviours by repeated visual observations throughout the stages of pairing, nest building, and fledging. Therefore, this technique can evaluate a broad community of species and be employed over large areas with reasonable time and effort (Gunn et al. 2000).

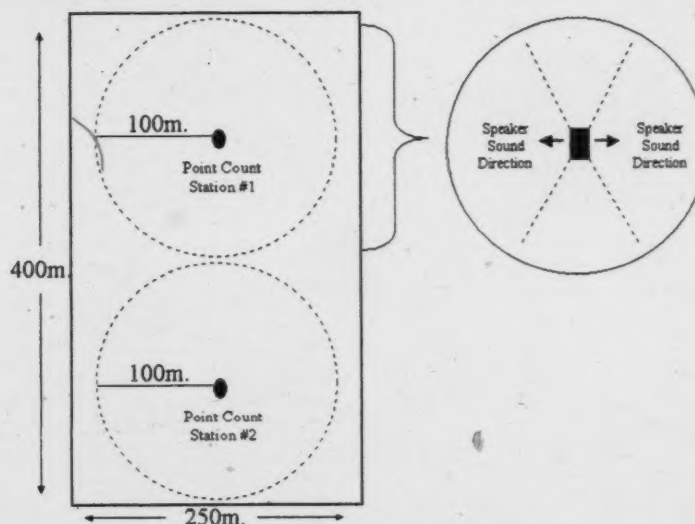
Two 100 m radius survey stations were located centrally within each of the 21 sites, for a total of 42 sampling stations. Surveys were comprised of three successive 5-minute intervals: 1) "pre-playback" passive point count, 2) call-playback, and 3) "post-playback" passive point count, totaling 15 minutes per survey. Observers passively documented all individual birds seen or heard within 50 and 100 m radii for the first 5 minutes. The observer then initiated the mobbing calls and recorded all individuals seen or heard, paying particular attention to direct interactions and specific behaviours that provided evidence of breeding activity (i.e. individuals carrying food or nesting material, territorial disputes, juveniles, etc.). The final 5 minutes was an additional point count (i.e. silent period) to ensure that inconspicuous species, or species that could not be heard over the sound of the call-playbacks, were documented.

Surveys were conducted during the second (2000) and third (2001) years post-fire and harvest. Each station was sampled repeatedly throughout the breeding season: 8 visits in 2000 (June 6 - July 14) and 7 visits in 2001 (June 7 - July 13). I felt it was important not to start the playback surveys until most birds had begun pairing, in case

regular mobbing broadcasts had negative effects on territory selection. Surveys were conducted between sunrise and 10:00 am (Ralph et al. 1993), only on days with little or no wind (~25km/h or <5 on the Beaufort scale) and no rain. Each observer sampled each station at least once, and survey times varied throughout the morning period, to account for observer bias and diurnal variation, respectively. To avoid sampling biases between stations, within each site (i.e. affecting the adjacent station), the first "point count" interval was conducted at both stations before either mobbing playback sequence was initiated.

I used Sony TCM-454VK hand-held cassette recorders and KOSS SA/35 portable amplified speakers to broadcast the call-playbacks. Speakers were placed perpendicular to the adjacent point count station, atop a tripod, approximately 1.3 m above ground (breast height) (Figure 3-2). Volume was set at a standardised auditory level within each of the 3 treatment types, resulting in the same volume setting for Controls and Leaves. Volume was slightly lower within Salvages due to less vegetative structure impeding sound travel. However, in all treatments, observers could not hear the mobbing broadcasts beyond the 100 m sampling radius.

Figure 3-2: Schematic showing the Black-capped Chickadee call-playback survey method used within each 10 ha site.



3.2.3 Selection of Focal Species for Testing Treatment Effects

To compare habitat quality among treatments, I selected 7 species, from the total 69 bird species documented from the surveys, that were relatively common and present within each of the different treatments: Black-capped Chickadee – BCCH, Brown Creeper – BR CR (*Certhis americana*), Red-breasted Nuthatch – RBNU (*Sitta canadensis*), Yellow-rumped Warbler – YRWA (*Dendroica coronata*), Mourning Warbler – MOWA (*Oporornis philadelphia*), Lincoln's Sparrow – LISP (*Melospiza lincolnii*), and White-throated Sparrow – WTSP (*Zonotrichia albicollis*). All 7 species were found within Control and Leave treatments, while LISP and WTSP were observed within Control, Leave, and Salvage treatments. In addition, these species represent a variety of nesting and foraging guilds that comprise a typical bird community within the boreal forest. BCCH, BR CR, and RBNU are resident species that nest in tree cavities or crevices. YRWA is a migrant species that nests higher in trees and typically forages within the canopy, gleaning the foliage for insects. MOWA, LISP, and WTSP are also migrants, but these species nest on the ground or low to the ground within a shrub or small tree. They also forage throughout the understory gleaning foliage and/or the leaf litter for insects (Ehrlich et al. 1988). Therefore, I assume these 7 species are a representative sub-sample of the bird community within each of the different treatment types.

3.2.4 Index of Relative Reproductive Activity

For each of the 7 focal species, I developed an index of relative reproductive activity that gauged the probability of breeding (i.e. larger index – increased confidence of breeding; smaller index – decreased confidence of breeding). All species observations were ranked from 1 to 4 according to individual behaviour that provided evidence of breeding activity (Table 3-1). The index incorporated observations gathered from each of the 3 intervals (entire 15-minute survey), to utilize all information available. The highest (maximum) ranked observation per species was selected from across survey intervals for each visit per sampling station. A mean of the "maximum" rank per species/visit was obtained by dividing sampling effort (# of visits) for individual years (2000 and 2001).

To eliminate the possibility of over sampling species within each site, indices from both survey stations within a site were averaged to obtain a single mean index for analyses.

Table 3-1: Rankings assigned to bird observations during point count and Black-capped Chickadee mobbing playback surveys.

Behaviour/Observation	Breeding Activity	Ranking
Singing/Countersinging male; Species Observed or Calling	Undetermined	1
Pair Observed; Territorial Dispute	Possible	2
Carrying Food; Distraction Display; Alarm Calling	Probable	3
Carrying Nest Material; Nest/Juvenile/Family Observed	Confirmed	4

3.2.4 Statistical Analysis

Call-Playback Effectiveness

Based on Gunn et al. (2000), I predicted that Black-capped Chickadee (BCCH) mobbing call-playbacks would provide increased evidence of breeding songbirds over point counts. Using the combined total of all species observations from 2000 and 2001, I conducted 1-sided *t*-tests comparing the proportion (%) of observations that ranked ≥ 2 within interval 1 and the proportion (%) of observations that ranked ≥ 2 within interval 2 (see Table 3-1 for ranks). To test whether treatment (habitat) type influenced the relative efficacy of this method, I conducted comparisons within each treatment first. The significance level for all tests was set at $\alpha = 0.05$.

Reproductive Activity Among Treatments

After testing whether relative reproductive activity differed between 2000 and 2001 within each treatment, I pooled yearly data to test for treatment effects. I used *t*-tests to compare reproductive indices between Control and Leave treatments for all 7 focal species combined, and for each species individually. I conducted 1-way ANOVA's to compare the combined, and individual indices, of LISP and WTSP among Controls, Leaves, and Salvage-logged treatments. Significant results were adjusted for multiple, pairwise comparisons using the Bonferroni post-hoc procedure (Winer et al. 1991). Non-parametric ANOVA's were used if assumptions of normality and variance could not be met to satisfy the use of parametric statistics. Alpha (α) was set to 0.10 for all statistical

tests to reduce the chances of committing a Type II error. Arguably, a Type II error in applied research can be more costly for conservation efforts than Type I errors (Peterman 1990; Smith 1995; Schmiegelow et al. 1997; Johnson 1999). Statistical procedures were carried out using SPSS 10.0.3 for Windows (SPSS Inc. 1999).

3.3 Results

3.3.1 Black-capped Chickadee Mobbing Playback Efficiency

As predicted, BCCH mobbing playbacks significantly increased the number of observations that provided evidence of breeding bird activity, compared to those of passive point count surveys (Figure 3-3). Per survey station/site, $34.0\% \pm 2.0$ of observations during the playback sequence (Interval 2) resulted in a ranked breeding behaviour ≥ 2 , compared to $13.4\% \pm 1.1$ of observations obtained from point counts (Interval 1) (1-sided $t_{41} = -8.309$, $p = <0.001$). Treatment type had no influence on the efficacy of the playback technique in acquiring observations of breeding activity. Playback intervals obtained significantly more observations that ranked ≥ 2 than point counts within each of the 3 treatments (Table 3-2).

Figure 3-3: Difference between point counts and Black-capped Chickadee mobbing playbacks in obtaining evidence of breeding activity from surveys conducted at Chip Lake. Graph indicates proportion of total bird observations ranking ≥ 2 (± 1 SE) (see Table 3-1 for behaviour rankings) for each survey method.

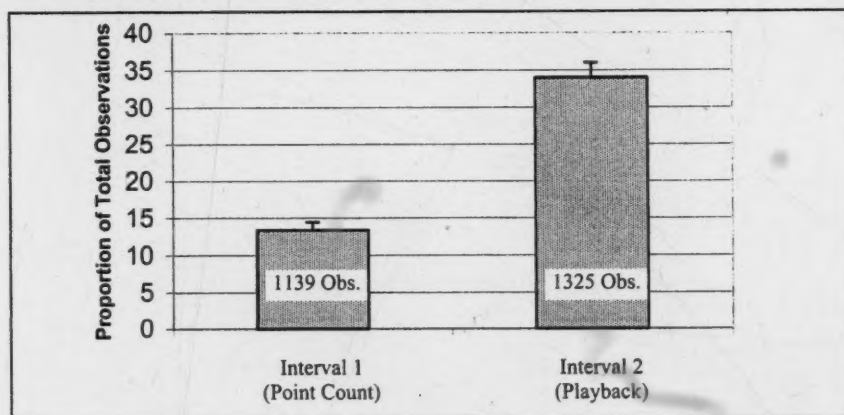


Table 3-2: Comparison between point counts (Interval 1) and Black-capped Chickadee mobbing playbacks (Interval 2) in obtaining observations of breeding birds, within each treatment type, during surveys conducted at Chip Lake. Bold type indicates significance at $\alpha = 0.05$.

Treatment ¹	Point Count (Interval 1)			Playback (Interval 2)			t	p
	% ²	SE	Total Obs.	% ²	SE	Total Obs.		
Control	11.9%	1.7	362	36.2%	2.3	606	-8.605	<0.001
Leave	11.1%	1.3	481	35.6%	3.1	573	-7.296	<0.001
Salvage	17.2%	2.1	296	30.4%	4.7	146	-2.588	0.019

¹ Control = unburned-unharvested; Leave = burned-unharvested; Salvage = burned-harvested

² Proportion of observations that ranked 2 or greater according to breeding behaviour rankings (see table 3-1).

Although BCCH mobbing playbacks provided higher proportions of total observations indicating breeding activity, it should be noted that individual species responded differently. Of the 7 focal species included in this study, BCCH, RBNU, and YRWA exhibited the strongest response to the mobbing sequence. BRGR exhibited an intermediate response, while MOWA, LISP, and WTSP responded poorly to the mobbing interval (Table 3-3). However, the third interval (post-playback silent period) proved to be quite effective in providing additional observations for MOWA, LISP, and WTSP. Observations during the third interval did not usually result in high ranked behaviours – being typically auditory – however, the mobbing sequence did appear to rouse these species and make them detectable. It is also possible that simply spending more time on a survey (sampling effort) increased the likelihood of detection. Appendix 10 provides a summary of the number of detections obtained for all species documented within each interval, including additional observations made during the third interval (silent period).

Table 3-3: Proportion (%) of total observations obtained during each Interval of the Black-capped Chickadee mobbing playbacks, for each of the 7 focal species, for surveys conducted at Chip Lake.

Species ¹	Total # of Obs.	Intervals			Resp. to Mobbing
		Interval 1 (Point Count)	Interval 2 (Playback)	Interval 3 (Silent Period)	
Black-capped Chickadee	172	30%	65%	5%	Good
Red-breasted Nuthatch	324	41%	49%	10%	Good
Yellow-rumped Warbler	248	46%	46%	8%	Good
Brown Creeper	84	51%	32%	17%	Med.
Mourning Warbler	308	75%	10%	15%	Poor
Lincoln's Sparrow	470	68%	18%	14%	Poor
White-throated Sparrow	566	68%	13%	19%	Poor

¹ See Appendix 2 for scientific names.

3.3.2 Relative Reproductive Activity Among Treatments

Although indices did appear to increase, no significant difference was detected in relative reproductive activity between the second (2000) and third (2001) year of surveys (Table 3-4). Subsequently, annual data were pooled to test for differences in reproductive indices among treatments. For the 7 species combined, Leaves supported a significantly higher reproductive index compared to Controls ($t_{26} = -1.830$; $p = 0.079$) (Figure 3-4a). The combined indices of LISP and WTSP also differed significantly among treatments ($F_{2,39} = 5.794$; $p = 0.006$). Indices were highest within Leaves but did not differ from Salvages; however, both Leave and Salvage indices were significantly higher than Controls ($p = 0.008$; $p = 0.042$, respectively) (Figure 3-4b).

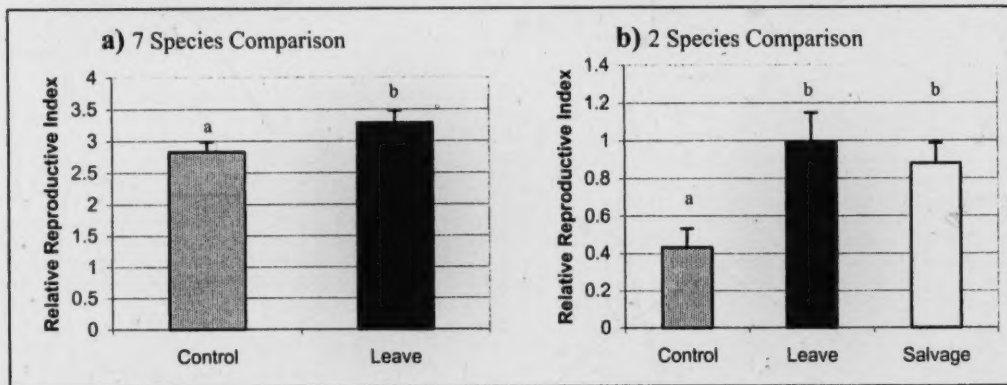
Table 3-4: Change in relative reproductive index within each treatment, between the 2nd (2000) and 3rd (2001) year of surveys conducted at Chip Lake.

Treatment ¹	2000		2001		<i>t</i>	<i>p</i>
	Index ²	SE	Index ²	SE		
Control	2.66	0.18	3.04	0.25	-1.383	0.192
Leave	3.00	0.32	3.57	0.18	-1.561	0.144
Salvage	0.75	0.13	1.01	0.17	-1.238	0.239

¹ Control and Leaves compare indices of Black-capped Chickadee, Brown Creeper, Lincoln's Sparrow, Mourning Warbler, Red-breasted Nuthatch, White-throated Sparrow, and Yellow-rumped Warbler; Salvages compare indices of Lincoln's and White-throated Sparrow (see Appendix 2 for scientific names).

² Derived from ranking individual breeding behaviours (see Table 3-1).

Figure 3-4: Comparison of relative reproductive activity (± 1 SE) among burned (Leave), unburned (Control), and salvage-logged (Salvage) treatments for surveys conducted at Chip Lake, 2-3 years post-fire. Treatments with different letters are significant at $\alpha = 0.10$. (a) Species include Black-capped Chickadee, Brown Creeper, Red-breasted Nuthatch, Mourning and Yellow-rumped Warbler, Lincoln's and White-throated Sparrow; (b) Species include Lincoln's and White-throated Sparrow.



YRWA was the only species that experienced significantly higher individual reproductive indices within Controls, while BRRCR, MOWA, and LISP experienced significantly higher indices within Leave treatments (Table 3-5). However, LISP indices showed significantly higher values within Salvages compared to Leaves (Table 3-6). There was no difference in relative reproductive activity for BCCH, RBNU, and WTSP between Leaves and Controls (Table 3-5), however WTSP indices were significantly higher within Leaves compared to Salvages (Table 3-6).

Table 3-5: Comparison of relative reproductive activity for individual species between burned (Leave) and unburned (Control) treatments (n = 14) during surveys conducted at Chip Lake, 2-3 years post-fire. Bold type indicates significance at $\alpha = 0.10$.

Species ¹	Controls		Leaves		t	p
	Index ²	SE	Index ²	SE		
Black-capped Chickadee	0.81	0.14	0.73	0.10	0.481	0.635
Brown Creeper	0.06	0.02	0.21	0.07	-2.139	0.048
Red-breasted Nuthatch	0.79	0.12	0.60	0.14	1.075	0.292
Mourning Warbler	0.18	0.06	0.41	0.08	-2.348	0.027
Yellow-rumped Warbler	0.55	0.08	0.35	0.09	1.772	0.088

¹ See Appendix 2 for scientific names.

² Mean "maximum" rank per visit (n=15) per site; Index derived from ranking individual breeding behaviours (see Table 3-1).

Table 3-6: Comparison of Lincoln's and White-throated Sparrow relative reproductive activity among burned (Leave), unburned (Control), and Salvage-logged (Salvage) treatments during surveys conducted at Chip Lake, 2-3 years post-fire. Bold type indicates significance at $\alpha = 0.10$.

Species ¹	Treatment ²			ANOVA ³	Multiple Comparisons	p ³
	Control	Leave	Salvage			
Lincoln's Sparrow	0.02	0.43	0.68	0.001	L > C	0.001
	(0.02)	(0.10)	(0.08)		S > C	<0.001
					S > L	0.084
White-throated Sparrow	0.41	0.57	0.20	0.011	C = L	0.570
	(0.10)	(0.07)	(0.07)		C = S	0.218
					S < L	0.009

¹ See Appendix 2 for scientific names

² Mean "maximum" rank per visit (n=15) per site (including standard error); Index derived from ranking individual breeding behaviours (see Table 3-1).

³ Values in italics derived from non-parametric ANOVA's.

3.4 Discussion

Due to increasing awareness of migratory songbird decline (Robbins et al. 1989; Rappole and McDonald 1994), and recent concerns over resident boreal birds (Schmiegelow and Monkkonen 2002), inquiry into habitat-specific productivity may be necessary to address concerns over habitat quality (Van Horne 1983; Vickery et al.

1992a; Purcell and Verner 1998). However, development of an efficient and accurate measure of obtaining reproductive success for avian communities has been a challenge.

Direct measures of nest success and productivity are undoubtedly more accurate than indirect indices based on behaviour; however, they are also extremely limiting in terms of observer effort required, species detected, and scale of application. The use of behavioural observations to estimate relative reproductive success for grassland birds has proven effective (Vickery et al. 1992b; Dale et al. 1997). Similar work conducted in forested habitats, however, has not met with the same success. Rangen et al. (2000) reported that indices based on reproductive behaviour and observations of fledglings are not feasible in forested areas, due to habitat complexity, unless species are very abundant and concentrate their activities below the lower canopy. I found using a response eliciting technique to obtain behavioural observations (Gunn et al. 2000) very helpful in acquiring observations of breeding activity. In forested environments, where visual observation of canopy species are limiting, the use of Black-capped Chickadee mobbing playbacks seems promising.

3.4.1 Efficacy of the Playback Technique

Like Gunn et al. (2000), I found BCCH mobbing call-playbacks resulted in a significantly higher proportion of observations of breeding activity than did passive observations. Behavioural evolution suggests that if similar species are present within different habitats, they should respond to a common stimulant (i.e. call-playback). Unless, however, the habitat types do not provide the same attributes (i.e. escape cover) to provide an individual the safety of "mobbing" a potential predator, or the species is unfamiliar with the stimulant (i.e. species does not recognize BCCH mobbing behaviour). Despite the very obvious structural differences between treatments types (see Appendices 7-9), I found treatment/habitat type (i.e. forest complexity) not to influence species response. These results corroborate Harrison (2002), who also reported no significant difference in the proportion of species observations obtained by BCCH playbacks, among various levels of forest retention, during an experimental harvesting study in northwestern Alberta, Canada. Application effectiveness in a variety of different studies reinforces the potential versatility of this methodology.

However, not all species within the local community responded to mobbing broadcasts, which indicates that some birds may not benefit from mobbing a predator (see Curio 1978 for details), or certain species that do not share similar predators, may not associate with other species alarm calls (Hurd 1996). MOWA, LISP, and WTSP did not directly respond to mobbing (i.e. make themselves visible to the observer), regardless of treatment. Because LISP and WTSP were the only species consistently abundant enough within salvage-logged treatments for comparison of habitat quality with other treatments, BCCH mobbing may not be a suitable option in obtaining a representative measure of breeding activity within this habitat type. Therefore, careful consideration should be given to individual response if specific species are of interest.

Focal species within Controls and Leaves responded to playbacks much more strongly. This is most likely due to the bird assemblage within each of these treatments more closely representing forest communities to which BCCH's belong. BCCH, RBNU, and YRWA all responded well, and were frequently observed, during the mobbing sequence. BRGR showed an intermediate response, whereas MOWA, as mentioned previously, did not respond directly to the mobbing playback. Many other species within these treatments also responded frequently to the mobbing sequence, including: Blue-headed Vireo (*Vireo solitarius*), Warbling Vireo (*Vireo gilvus*), Golden-crowned Kinglet (*Regulus satrapa*), Ruby-crowned Kinglet (*Regulus calendula*), Tennessee Warbler (*Vermivora peregrina*), and Swainson's Thrush (*Catharus swainsoni*) (see Appendix 10). However, additional species were not included in this study because they rarely occupied both habitats, and thus could not be directly compared.

This study suggests that BCCH mobbing call-playbacks can be effective, specifically in forested habitats, in providing additional observations of reproductive activity for a suite of bird species. Call-playbacks can provide similar observations that direct behavioural monitoring achieves, although there is far less effort required of the observer. Furthermore, there is less risk of nest abandonment or predation of eggs and young (Martin and Geupel 1993), caused by observer carelessness, when direct nest observations can be avoided. In terms of a census methodology for sampling forest bird communities, call-playbacks may be an ideal complement to point counts for detecting more elusive and secretive birds, especially given that some species wane in singing

frequency throughout the breeding season (Best 1981; Skirvin 1981). Both Gunn et al. (2000) and Harrison (2002) reported similar results, in that BCCH mobbing call-playbacks increased the number of species observations, and combined with point counts, more thoroughly census an area. Results from this study also indicate that call-playbacks provided additional information to the observer that may have been missed by strictly passive observation.

3.4.2 Relative Habitat Quality of Burned and Unburned Forest

I created a sub-sample of the boreal forest bird community by combining the reproductive indices of BCCH, BRCR, RBNU, MOWA, and YRWA. Although proportionally this sample is slightly biased towards resident species (BCCH, BRCR, and RBNU) compared to migrants (MOWA and YRWA), increasing concerns regarding resident species within the boreal forest (see Schmiegelow and Monkkonen 2002) warrant a more detailed examination into their breeding activities.

The combined sample-community experienced significantly higher reproductive indices within burned (Leave) forest compared to unburned (Control) forest. BCCH and RBNU did not differ in reproductive activity between Leaves and Controls, indicating both habitats tend to support these species equally well. BRCR, a species for which very little is known in Alberta (Alberta Sustainable Resource Development 2000), experienced significantly higher reproductive indices within Leaves, 2-3 years post-fire. MOWA also experienced higher reproductive activity within Leaves, whereas YRWA experienced higher indices within the Controls. These results suggest that burned, older forests provide at least a similar level of habitat quality as unburned older mixedwood forest, very early in the "new" seral stage. This is particularly significant as the bird community found breeding within early successional post-fire forests more closely represents that within older growth forests (see chapter 2). The same could not be said for burned forests that were salvage-logged.

Reproductive indices for individual species mirrored what their natural history characteristics suggest. BRCR's tend to nest under loosened bark that lifts from the trunks of trees or snags. Most BRCR nests that were located as part of a complementary

experiment (see chapter 4), were found within burned sites, and located underneath loosened bark caused by the effects of fire. Anecdotally, it should also be noted that some nests rested precariously underneath intensely burned bark that high winds dislodged, resulting in nest failure; over 3 years, at least two such cases were observed. MOWA's prefer moist, shrubby understory vegetation (Ehrlich et al. 1988), which was characteristic of early-successional burned forests in this study (see Appendices 7-9). Not surprisingly, YRWA, a species which nests primarily in coniferous trees and forages in the mid to upper canopy (Ehrlich et al. 1988), experienced higher reproductive indices within unburned forest. Only sites burned at relatively low intensity supported YRWA's; most burned sites within this study contained neither an upper canopy nor had many live conifer trees.

These results differ from White et al. (1999), who monitored bird abundance and reproduction through direct nest monitoring, within burned and unburned pine forests of Georgia, USA. Similar to this study (see Chapter 2), White et al. (1999) found that more species preferred burned sites to unburned sites; however, White et al. (1999) report low productivity estimates within the burned areas. Nevertheless, due to very low numbers of nest locations within unburned sites, White et al. (1999) could not statistically compare productivity between habitat types. They do, however, suggest nest predation may be the cause for low productivity within the burned sites.

I feel BCCH mobbing call-playbacks complement passive census techniques in providing a level of reassurance regarding habitat quality. However, I do not advocate this methodology as a substitute for direct monitoring of avian productivity where single species concerns are a priority. BCCH playbacks cannot quantify nest predation or brood parasitism. American Kestrels (*Falco sparverius*), which are known to prey on small passerines, were frequently observed within burned areas and nest locations were found in tree cavities created by the fire. Brown-headed Cowbirds (*Molothrus ater*) were also observed throughout the study area; however, no evidence of brood parasitism was found in any of the treatments. Where direct concerns for single species exist, more intensive techniques may be necessary to better understand habitat dynamics.

3.4.3 Relative Habitat Quality of Salvage-logged Areas

The only species of substantial quantity within salvage-logged areas, that were also present in the other 2 treatments, were LISP and WTSP. LISP was the only species to show significantly higher relative reproductive activity within salvage-logged areas, compared to both Leave and Controls. These results are not surprising, considering LISP was not very abundant within Controls. Further, LISP's preference to nest on the ground within moist thickets and/or meadows (Ehrlich et al. 1988), quite typical of Salvages and to a lesser extent Leaves, is not characteristic of older forests. WTSP indices were lowest within salvage-logged areas, and while no different from Controls, were significantly lower than Leaves. These results corroborate Hannah (2001), who studied habitat quality for White-throated Sparrows in north-central Alberta, Canada (~300 km northeast of this study site), finding WTSP reproductive indices and nesting success to be higher within interior forest than in 5-6 year old cutblocks. Results from this study suggest that burned forests also provide quality breeding habitat for WTSP's.

While not to diminish their intrinsic value, LISP and WTSP are not species of special concern with regards to forest management practices, and as previously mentioned, were not good candidates for responding to the BCCH mobbing playbacks. Perhaps, to some extent, the absence of certain bird species and the distinct difference in bird communities (see Chapter 2) between burned and salvage-logged areas, indicate a difference in habitat quality. Where a more representative assessment of the bird community within salvage-logged areas is of interest, perhaps methodologies successfully employed for open, grassland areas (see Vickery et al. 1992b; Dale et al. 1997) would be more appropriate.

3.5 Conservation Implications

The mixedwood boreal region supports the highest diversity of breeding songbirds in Canada. Quality breeding habitat is vital to species persistence and salvage-logging burned forests substantially reduces the value of this habitat for forest birds. Recognising that many bird species utilize burned forests immediately post-fire, and that

these areas provide quality breeding habitat, is necessary to ensure that reproductive sources are maintained within the boreal region, through landscape-level planning.

Five of the 7 species from the sample bird community in this study exhibited similar, or had higher, indices of relative reproductive activity within older-aged burned forest, compared to unburned forest of the same age. These results indicate that burned forests can sustain valuable breeding habitat, utilized by both migrant and resident bird species found in older-aged, unburned forests, typically regarded as "high quality" habitat. The Brown Creeper, a species that appeared to benefit from the availability of this early-seral habitat, is of "undetermined" status in Alberta due to the lack of information known about this species. Understanding preferred habitat requirements for this, and other species, is a timely concern, considering the increasing level of resource development in the boreal region.

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Chapter 4: Dynamics of the Cavity-Nesting Bird Community: Nest Webs and Nest Selection

4.1 Introduction

Cavity-dependent bird species are typically associated with older, unmanaged forests that contain larger diameter trees, many in a state of fungal decay, including dead trees (snags). Numerous studies have demonstrated the importance of these characteristics to cavity-nesting birds, for reasons such as foraging, nesting, and roosting (e.g., Sedgwick and Knopf 1986; 1990; Runde and Capen 1987; many others). Cavity-nesting birds within regions of Alberta's boreal forest, account for roughly 1/3 of the total forest bird community. Brawn et al. (1982) suggested that by maintaining the cavity-nesting bird community, a more diverse songbird community would also be supported. Furthermore, it may be possible to predict forest bird diversity by the number of cavity-nesting species (Mikusinski et al. 2001).

While protection and maintenance of critical habitat features necessary for species persistence is typically a conservation priority, identifying critical features and understanding the long-term implications of their loss, is a substantial challenge. Martin and Eadie (1999) proposed a unique approach to examining cavity-nesting bird communities, including linkages and interactions between cavity-nesting guilds in a hierarchal structure, analogous to a food web. Martin and Eadie (1999) propose a "nest web" structure comprised of 3 species guilds: (1) Primary cavity nesters (PCN) – species that can readily excavate their own cavity for nesting (e.g., woodpeckers (*Picidae*)), (2) Weak cavity nesters (WCN) – species capable of excavation through soft or decayed sapwood, but which may also use existing cavities for nesting (e.g., chickadees (*Paridae*) and nuthatches (*Sittidae*)), and (3) Secondary cavity nesters (SCN) – species dependent on previously excavated or naturally occurring cavities for nesting (e.g., Tree Swallow (*Tachycineta bicolor*), House Wren (*Troglodytes aedon*), American Kestrel (*Falco sparverius*), etc).

In the hierarchy of a nest web, cavity-nesters require cavities (i.e. food/prey in a food web) as a central resource (Martin and Eadie 1999). Within forested systems,

cavities come exclusively from trees and/or snags, with primary- and weak cavity-nesters being primary consumers of the resource (i.e. second trophic level in a food web). Secondary cavity-nesters are considered secondary consumers if they reuse previously excavated cavities by PCN's or WCN's. However, although SCN's cannot excavate their own cavities, they may also be considered primary consumers if they utilize naturally occurring cavities (e.g., those created by fire).

I attempt to identify species and/or guild associations by simulating a nest web structure with cavity nests located over 3-years (1999-2001), from 3 different habitat types: (1) ~100 year-old deciduous-dominated mixedwood forest, (2) ~100 year-old deciduous-dominated mixedwood forest burned by wildfire, and (3) ~100 year-old deciduous-dominated mixedwood forest burned by wildfire and salvage-logged. Because the majority of cavity-nesting birds in this region utilize snags over live trees for cavity nests, I also wanted to know if cavity-nesters selected nest locations within habitats based on snag density (i.e. 3rd order selection; Johnson 1980). At a finer scale, 4th order selection (Johnson 1980), I wanted to know if cavity-nesters selected nest-snags, within a nest location, according to snag diameter and height.

Snag availability is not the only factor responsible for nest selection; food resources, predation, and competition may also influence this process (see Swallow et al. 1986). However, from a forest harvesting perspective, snags are the most recognizable and manageable feature. If a healthy and diverse cavity-nesting bird community can be maintained among industry development, this may also benefit many other songbirds (i.e. a type of coarse-filter approach).

4.2 Methods

4.2.1 Study Area

This study was conducted in the foothills natural region (Alberta Sustainable Resource Development 1996) of west-central Alberta, within the Weyerhaeuser (Edson) Forest Management Agreement. During the spring of 1998, an estimated 14,000 ha of forested land, primarily black spruce (*Picea mariana*) and tamarack (*Larix laricina*)

bottomlands, were burned by wildfire. Approximately 2,200 ha consisted of older deciduous-dominated upland forest, sub-dominated by white spruce (*Picea glauca*). This forest was deemed merchantable for harvest, and subsequently salvage-logged in the winter of 1998/1999. This region also has a prior human footprint due to the historical high-grading of white spruce. Furthermore, fire suppression techniques (e.g., fire retardants) employed to contain the wildfire ultimately limited its extent.

Twenty-one study sites, 10 ha in size (250 x 400 m), were selected from a 10 x 10 km township (Twp: 51, Rge: 09, W5M) (Figure 4-1). Seven sites were established as "Controls" (unburned and unharvested) and were located outside the burn perimeter in continuous forest. Seven sites were designated "Leaves" (burned and unharvested) and seven were designated "Salvages" (burned and harvested). All sites were selected from older aged (≥ 100 years) deciduous-dominated mixedwood stands, comprised primarily of trembling aspen (*Populus tremuloides*), balsam poplar (*Populus balsamifera*), and white spruce. Understory vegetation consisted of two dominant "tall shrub" species (obtaining heights ≥ 3 m): willow (*Salix* spp.) and green alder (*Alnus crispa*). Other dominant shrub species included bracted honeysuckle (*Lonicera involcrata*), prickly rose (*Rosa acicularis*), currant and gooseberry, (*Ribes* spp.), and low-bush cranberry (*Viburnum edule*).

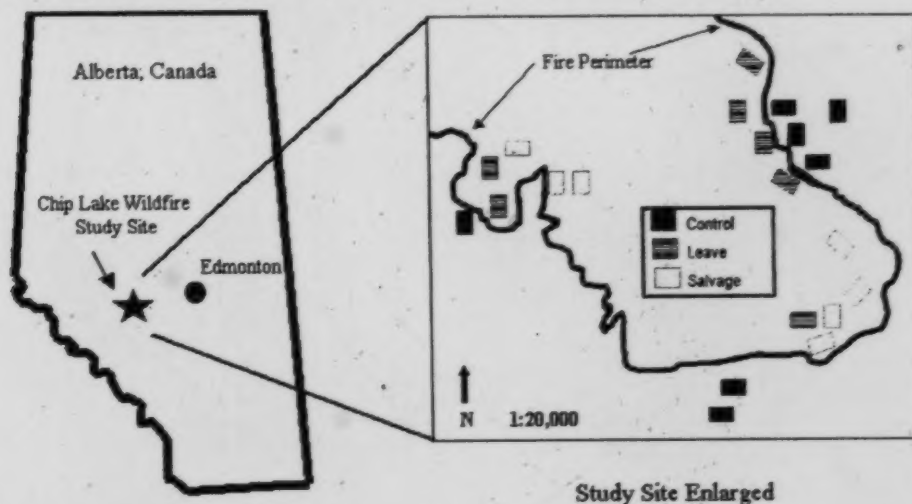
To the extent possible, treatment and control sites were spatially interspersed, separated by a minimum of 200 m. Sites were considered spatially independent for the use of inferential statistics (Hurlbert 1984).

4.2.2 Bird Surveys

Abundance of cavity-nesting species was obtained during the breeding season using the spot mapping method (International Bird Census Committee (IBCC) 1970). Surveys were conducted between the last week of May and the second week of July, during 1999, 2000, and 2001. All species seen or heard were recorded and each site was sampled a minimum of 7 times per season. All surveys were conducted between sunrise and 10:00 am on days with little wind, (~ 25 km/h or < 5 on the Beaufort scale), and no rain. To standardize sampling effort across treatments, observers spent 1.5 - 2 hrs per

site, recording the exact duration of each survey. To minimize observer bias and diurnal variation, all observers surveyed each site at least once per year and survey start times varied throughout the morning period.

Figure 4-1: Location of the study site within Alberta, Canada and the experimental research design within the southeast corner of the 1998 Chip Lake wildfire.



4.2.3 Nest Searches

Nest searches consisted of observers revisiting sites after spot mapping, so they could use the location information gathered earlier to refine their search area within each 10 ha site. Some incidental nest locations were found during the spot mapping surveys. Observers spent a maximum of 1 hour in search of a cavity-nesting individual, and would then observe that individual for a maximum of 30 minutes, or until the nest was found. If time permitted, the observer would then resume searching for another cavity-nesting individual. To standardise sampling effort among treatments, observers limited their "search" time to 1 hour per 10 ha site, or a total combined search and observation time of 1.5 hours, per visit. If continued observations of cavity-nesting species occurred during subsequent spot-mapping surveys, observers would continue to search for nests.

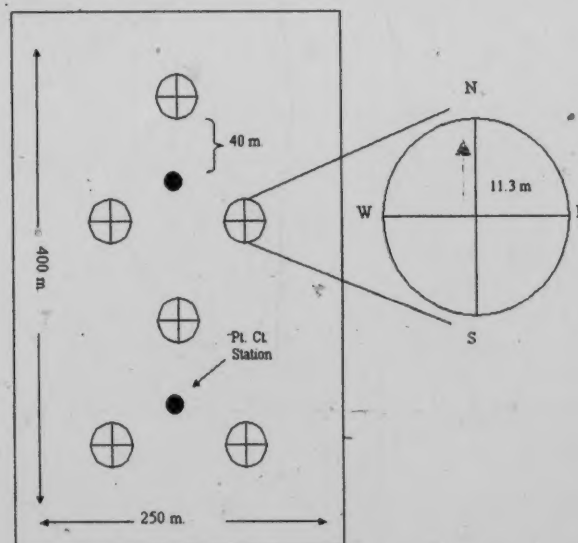
Due to limited sample sizes of cavity nests within the 10 ha sites, observers also searched areas of similar size and stand composition outside the 10 ha sites, after I was confident most cavity-nests had been located within the sites. The same search protocols

were used when searching areas outside of the 10 ha sites, including standardising sampling effort among habitat types.

4.2.4 Habitat Characteristics

Habitat characteristics were collected from circular, 11.3 m radius plots (0.04 ha). Plots were established around each cavity nest (i.e. nest-tree as the origin), to document nest-tree and snag characteristics immediately adjacent to the nest-tree, considered the "nest location". Plots were also situated systematically within each 10 ha study site to measure snag attributes considered randomly available to cavity nesting birds. Plots were situated 40 m distance at 0, 120, and 240 degrees from each of two centrally located points within each 10 ha site, totaling 6 plots per site (Figure 4-2) and equaling 126 survey plots per year. For all snags >12 cm diameter-at-breast height (DBH), actual DBH, snag height using a clinometer, and snag species (when possible) were recorded. All snags <12 cm DBH were simply tallied. Many other vegetative metrics were collected, but not included in these analyses. All measurements were taken from mid to late July of each year.

Figure 4-2: Design layout of the vegetation plots within each 10 ha site where snag characteristics were measured. Layout of the nest plots was identical to that of a single vegetation plot.



4.2.5 Statistical Analyses

Species- and Guild-Level Relationships

I schematically examined the flow and turnover of all cavity nests found from 1999 to 2001, within a web structure, to identify potential species linkages. Correlation analyses were conducted between the 3 guilds of cavity-nesting species (PCN, WCN, and SCN), using abundance data obtained from spot-mapping surveys, to examine if relationships occurred at the guild level. Abundance estimates were derived for each year by summing the number of individuals per site, separated into guilds, and adjusted for sampling effort by dividing estimates by the sum of survey time spent per site (i.e. $\text{abundance} = \text{number of individuals} / \text{hour of survey time} / \text{site}$). Because the study system was quite dynamic, due to temporal changes associated with post-fire succession, tests were conducted separately for each year (1999-2001), and for all years combined. To corroborate findings, correlation analysis was also applied to actual nests found among guilds.

I predicted PCN's and SCN's would be positively correlated (1-sided tests), based on SCN's requirement for existing cavities, either natural or previously excavated. No other predictions between guilds were made, due to potential competition for nest sites.

Nest Selection

Various statistical methods have been employed to test resource selection, but many require categorical data (Allredge and Ratti 1986; 1992). Continuous data can be divided into categories, but divisional break points need to be justified, as these can influence the results (Johnson 1980). At a coarse spatial scale, I used logistic regression to test if cavity-nesting birds select "nest locations" relative to snag density, within a given habitat type. Nest location in this study is defined as the immediate (0.04 ha) area surrounding the nest-snag. Because different species can have different selection processes, I felt pooling species data into the 3 previously mentioned guilds (PCN, WCN, and SCN) would provide a broad interpretation of patterns. I conducted independent tests for each guild.

I used only those nest locations found within the 10 ha study sites for nest selection analyses. To generate the binary data necessary for logistic regression, I combined the data from all systematically acquired plots (value of 0) for each year, into a single file with "nest locations" (value of 1), by guild. By pooling treatments, the data represent a range of conditions, from low snag density (salvage-logged treatments) to high snag density (burned and unharvested treatments). The smallest diameter snag used for a nest in this study was 12.5 cm DBH, and I deemed this a reasonable biological cut-off point, whereby snags <12.5 cm DBH were not considered available for use by cavity-nesting species in this study. I analyzed each year separately due to annual variation in available snag densities, especially within burned/unharvested treatments (trees continued to die with time post-fire).

At a finer spatial scale, I tested if cavity-nesting birds (according to guild) selected "nest-snags" within their nest location according to specific snag attributes. Attributes of special interest were snag DBH and snag height. Using logistic regression, I compared the paired values of DBH and height for nest-snags (value of 1), by guild, with DBH and height of available snags within the nest location (value of 0). I pooled the data for all treatments and years, considering available snag attributes were measured within nest locations that were selected by species in each year. Therefore, this analysis accounts for annual variation. However, at this spatial scale, I felt that snag availability should be more refined. I classified only those snags larger than 14.5 cm DBH as available to PCN's, based on their larger body size. Snags larger than 12.5 cm DBH were classified as available to SCN's and WCN's.

All logistic regression analyses were tested for non-linearity using the Box-Tidwell test and for collinearity using tolerance statistics (Menard 1995). I looked for outliers in the data using studentized residuals, leverage and dbeta diagnostics (Menard 1995). Logistic regression tests were interpreted using the odds ratio, calculated by $\text{Exp}(B) - 1$ from SPSS Version 10.0.3 for Windows (SPSS Inc. 1999), which indicates the percent change in the probability of nest use, associated with a 1% change in snag characteristics. To corroborate the findings from logistic regression, t-tests were used to compare the mean of used snag attributes (density, DBH, and snag height) to that available. T-tests also provided confidence intervals to specify a range of selected snag-

characteristics for each species guild. Alpha (α) was set to 0.10 for all statistical tests, to reduce the chances of committing a Type II error. Arguably, a Type II error in applied research can be more costly for conservation efforts than Type I errors (Peterman 1990; Smith 1995; Schmiegelow et al. 1997; Johnson 1999). Finally, because 97.3% of all cavity nests were located in deciduous snags, all tests were repeated with and without coniferous snags (in addition to the existing "availability" criteria), as recommended by Thomas and Taylor (1990).

4.3 Results

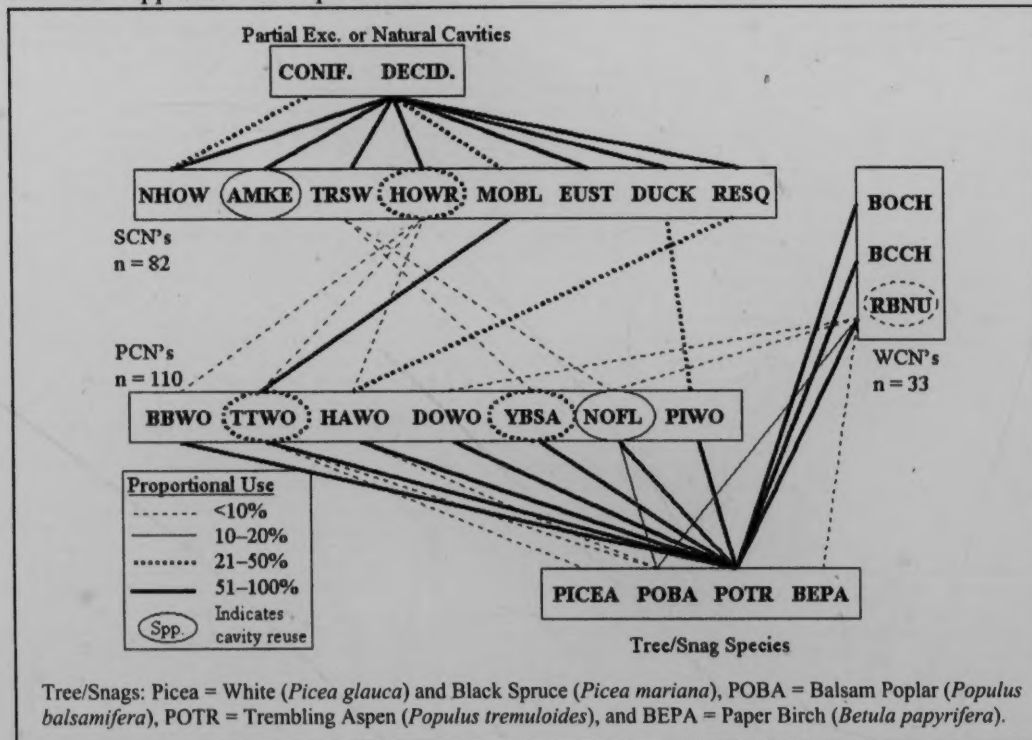
4.3.1 Species Relationships

We located 197 active cavities from 1999 to 2001 (Appendix 11). As 28 cavities were reused over the 3 years, species turnover was monitored from 225 cavity nests. We found 110 nests from 7 PCN species: Black-backed (*Picoides arcticus*), Three-toed (*P. tridactylus*), Hairy (*P. villosus*), Downy (*P. pubescens*), and Pileated Woodpecker (*Dryocopus pileatus*), Northern Flicker (*Colaptes auratus*) and Yellow-bellied Sapsucker (*Sphyrapicus varius*). Thirty-three nests from 3 species of WCN were found: Black-capped Chickadee (*Parus atricapillus*), Boreal Chickadee (*P. hudsonicus*), and Red-breasted Nuthatch (*Sitta canadensis*). Eighty-two SCN cavities were located from 7 avian and 1 mammal species: Tree Swallow (*Tachycineta bicolor*), House Wren (*Troglodytes aedon*), Mountain Bluebird (*Sialia currucoides*), European Starling (*Sturnus vulgaris*), Northern Hawk-Owl (*Surnia ulula*), American Kestrel (*Falco sparverius*), a duck species (*Bucephala clangula* or *B. albeola*), and Red Squirrel (*Tamiasciurus hudsonicus*).

PCN and WCN species almost exclusively (97.3%) excavated cavities in deciduous trees or snags, while 78.6% of all cavity nests were located in snags. Three-toed woodpecker was the only species that excavated nest cavities within conifer snags. Excluding all cavities from Yellow-bellied Sapsuckers, which nested almost exclusively in live trembling aspen (*Populus tremuloides*) with signs of fungal decay (conks present), resulted in 91.9% of all nests being located within snags. WCN species tended to excavate their own nest cavities into snags that exhibited later stages of decay, with the

exception of 2 Red-breasted Nuthatches that reused a Downy Woodpecker and Northern Flicker cavity (Figure 4-3). Interestingly, relatively few SCN species used fresh PCN nest cavities from previous years, during the 3 years of monitoring. The majority of SCN nests were located in either natural cavities or old partial excavations from PCN foraging or roosting; subsequently, no confident species identification could be made to the source of partial excavations. The use of natural cavities within this system is not surprising, considering most of the nests were located in areas 1-3 years post-fire, with fire being an important source of natural cavities. Mountain Bluebird was the only species to show an affinity towards one particular PCN cavity, the Three-toed Woodpecker. Three species of PCN, 2 species of SCN, and 1 species of WCN reused the same cavity excavated the year previous, although no proof could be provided that it was the same pair.

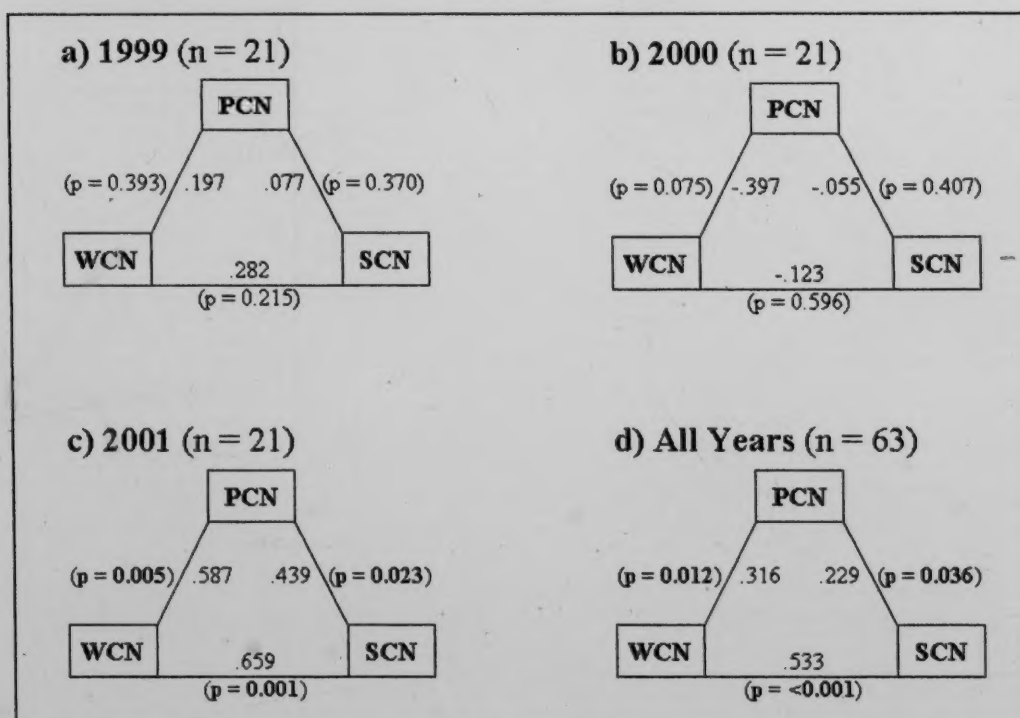
Figure 4-3: Nest web structure (after Martin and Eadie 1999) illustrating the flow and turnover of cavity resources among nesting guilds. Lines and ellipses indicate the proportion of cavities used from the level below or reused by the same species, respectively. Natural cavities flow downward. Data was obtained from unburned, burned, and burned/harvested habitats the first 3-years of monitoring (1999-2001) post-fire. See Appendix 1 for species codes.



4.3.2 Guild Relationships

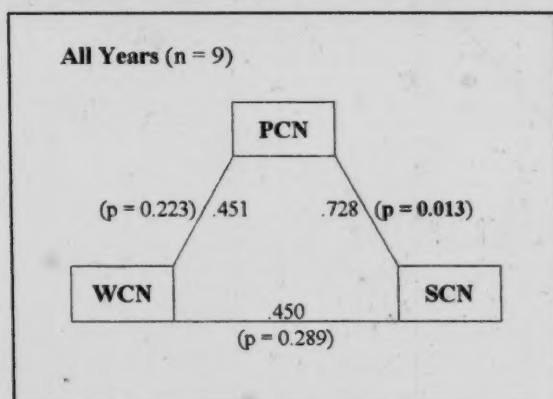
Correlation analysis for cavity-nester abundance indicated positive and significant relationships between all 3 guilds over the entire 3-years (Figure 4-4d). However, because the natural disturbance component to this study (wildfire) created a dynamic early-successional habitat type, independent within-year tests indicate a time lag effect (Figure 4-4a-c). Annual relationships were non-significant until the third year (2001).

Figure 4-4: Pearson correlations between abundances of primary cavity-nesters (PCN), weak cavity-nesters (WCN), and secondary cavity-nesters (SCN), collected at Chip Lake from 1999 (1st year post-fire) to 2001 (3rd year post-fire).



Nest correlations between guilds and across years corroborated a significant positive relationship between PCN and SCN's, while relationships between other guilds were non-significant (Figure 4-5). All relationships for independent years were consistently positive but not significant. However, insufficient data (zero WCN nests) in 1999 left relationships between WCN's inconclusive.

Figure 4-5: Pearson correlation of all cavity nests among years between primary cavity-nesters (PCN), weak cavity-nesters (WCN), and secondary cavity-nesters (SCN), collected at Chip Lake from 1999 (1st year post-fire) to 2001 (3rd year post-fire).



4.3.3 Selection of Nest Locations: Coarse-Scale

Primary and secondary cavity nesters used "nest locations" (0.04 ha surrounding nest) with a significantly higher mean snag density (≥ 12.5 cm DBH), compared to that available to them in each year (Table 4-1). PCN and SCN's also selected nest locations with a significantly higher density of deciduous snags. No WCN nests were located in 1999. However, WCN's did not use nest locations with either a higher overall snag density (≥ 12.5 cm DBH), or deciduous snag density, compared to that available to them in either 2000 or 2001 (Table 4-1).

Table 4-1: Difference in snag densities (≥ 12.5 cm DBH) between nest locations (0.04 ha surrounding nest tree) and densities deemed randomly available to species within specific nesting guilds, for surveys conducted at Chip Lake from 1999-2001. Bold type indicates significance at $\alpha = 0.10$.

Guild ¹	Year	Nest Location		Available ²		<i>t</i>	<i>p</i>
		Density	SE	Density	SE		
All Snags							
PCN	1999	17.1	2.28	4.7	0.56	6.388	<0.001
	2000	10.1	1.38	5.1	0.62	3.776	<0.001
	2001	10.8	2.49	5.6	0.67	2.551	0.012
WCN	1999	----	----	----	----	----	----
	2000	5.1	1.13	5.1	0.62	-0.022	0.983
	2001	8.2	2.34	5.6	0.67	1.024	0.308
SCN	1999	11.2	2.63	4.7	0.56	2.425	0.013
	2000	12.2	1.27	5.1	0.62	5.397	<0.001
	2001	13.9	2.57	5.6	0.67	3.126	0.007
Deciduous Snags							
PCN	1999	6.3	1.05	3.0	0.31	3.167	0.002
	2000	5.5	0.95	3.2	0.33	2.926	0.025
	2001	4.7	1.31	3.6	0.36	0.998	0.320
WCN	1999	----	----	----	----	----	----
	2000	4.2	0.99	3.2	0.33	0.884	0.400
	2001	5.7	1.21	3.6	0.36	1.478	0.142
SCN	1999	5.8	2.26	3.0	0.31	1.420	0.024
	2000	5.7	0.75	3.2	0.33	3.482	0.001
	2001	5.9	1.25	3.6	0.36	1.936	0.055

¹ PCN = Primary cavity-nesters; WCN = Weak cavity-nesters; SCN = Secondary cavity-nesters (see introduction).

² Densities derived from mean values obtained from 126, 0.04 ha plots, systematically located through 21 study sites for which nests were located.

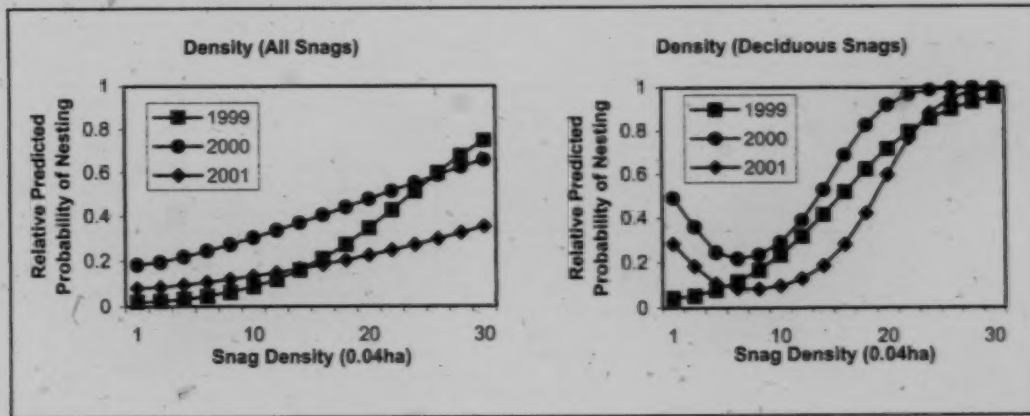
Logistic regression indicated "all snag" density and "deciduous snag" density as significant variables in predicting PCN nest selection, in all 3 years (Table 4-2). Non-linear forms for deciduous snag density were present in 2000 (Box-Tidwell test, chi-squared = 3.144, df = 1, $p = 0.05$) and 2001 (Box-Tidwell test, chi-squared = 4.171, df = 1, $p = 0.05$), therefore the variable (Density)ln(density) was retained in the model for each year. In all 3 years, the relative predicted probability of PCN's to select a nest location generally increased with snag density (Figure 4-6). The relative predicted probability of PCN's to select nest locations also tended to asymptote at approximately 25 deciduous snags/0.04 ha. The initial downward trend for 2000 and 2001 PCN's, in the

deciduous model, is likely caused by pooling of data. We began locating Northern Flicker nests within salvage-logged areas during the second (2000) and third (2001) year post-fire. Many of the nests were in single snags among barren salvage-logged sites, as indicated by the trend. In each model for "all snag" density, 92.7%, 73.7%, and 87.4% of the locations were correctly classified as used or available and 41.2%, 6.6%, and 5.3% of the variation between used and available densities were accounted for by the models in 1999, 2000, and 2001, respectively. Deciduous snag models correctly classified 90.6%, 69.1%, and 84.8% of the locations as used or available and models accounted for 11.0%, 7.3%, and 6.4% of the variation between used and available densities in 1999, 2000, and 2001, respectively. Models for "all snag" density in 2000 and 2001 indicated a lack-of-fit to the data (Hosmer-Lemeshow $p = <0.001$ and $p = 0.005$, respectively).

Table 4-2: Primary cavity-nester (coarse-scale) logistic regression models describing the importance of snag density (≥ 12.5 cm DBH) for the selection of nest locations. Odds ratio indicates the percent change in the probability of nesting associated with a 1% change in the variable.

Variables	B	SE	p	Odds Ratio	Variables	B	SE	p	Odds Ratio
1999 "All Snags" n=138, chi-sq= 23.739, df=1, p=<0.001					1999 "Deciduous Snags" n=138, chi-sq=8.086, df=1, p=0.004				
Constant	-4.073	0.651	<0.001		Constant	-3.299	0.531	<0.001	
Density	0.172	0.039	<0.001	18.80%	Density	0.211	0.075	0.005	23.50%
2000 "All Snags" n=171, chi-sq=12.274, df=1, p=<0.001					2000 "Deciduous Snags" n=171, chi-sq=10.791, df=2, p=0.005				
Constant	-1.568	0.248	<0.001		Constant	1.022	0.696	0.142	
Density	0.074	0.022	0.001	7.70%	Density	-1.06	0.391	0.007	-65.3%
					(Density)	0.377	0.138	0.006	45.80%
2001 "All Snags" n=143, chi-sq=5.217, df=1, p=0.022					2001 "Deciduous Snags" n=143, chi-sq=5.099, df=2, p=0.078				
Constant	-2.499	0.366	<0.001		Constant	0.139	0.886	0.875	
Density	0.063	0.027	0.018	6.50%	Density	-1.066	0.476	0.025	-65.6%
					(Density)	0.36	0.159	0.024	43.30%
					ln(density)				

Figure 4-6: Primary cavity-nester (coarse-scale) relative predicted probability of selecting a nest location relative to snag density (≥ 12.5 cm DBH).

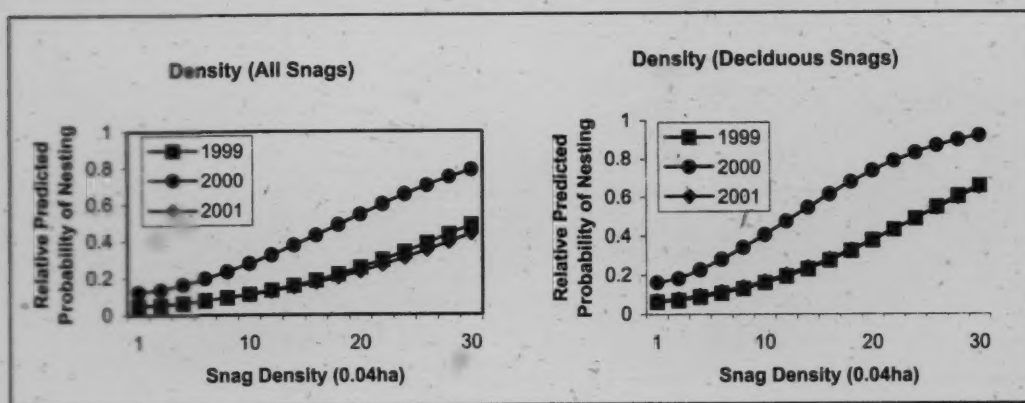


Logistic regression did not find snag density, or the density of deciduous snags, significant explanatory variables for WCN's in either 2000 or 2001. Logistic regression identified "all snag" and "deciduous snag" density significant variables in all models for SCN's (Table 4-3). For snags ≥ 12.5 cm DBH, the relative predicted probability of SCN's to select a nest location generally increased with snag density and deciduous snag density, in all 3 years (Figure 4-7). Models for "all snag" density correctly predicted 91.2%, 75.2%, and 89.3% of the locations as used or available and accounted for 10.6%, 14.7%, and 12.9% of the variation between used and available densities in 1999, 2000, and 2001, respectively. Models for "deciduous snag" density correctly predicted 92.0%, 75.8%, and 90.0% of the locations as used or available and accounted for 5.0%, 6.3%, and 3.7% of the variation between used and available densities in 1999, 2000, and 2001, respectively.

Table 4-3: Secondary cavity-nester (coarse-scale) logistic regression models describing the importance of snag density (≥ 12.5 cm DBH) for the selection of nest locations. Odds ratio indicates the percent change in the probability of nesting associated with a 1% change in the variable.

Variables	B	SE	p	Odds Ratio	Variables	B	SE	p	Odds Ratio
1999 "All Snags" n=138, chi-sq= 7.770, df=1, p=0.005					1999 "Deciduous Snags" n=138, chi-sq=3.854, df=1, p=0.050				
Constant	-3.093	0.463	<0.001		Constant	-2.823	0.417	<0.001	
Density	0.101	0.035	0.004	10.6%	Density	0.116	0.057	0.044	12.3%
2000 "All Snags" n=165, chi-sq=23.091, df=1, p=<0.001					2000 "Deciduous Snags" n=165, chi-sq=10.639, df=1, p=0.001				
Constant	-2.087	0.302	<0.001		Constant	-1.788	0.285	<0.001	
Density	0.113	0.025	<0.001	11.9%	Density	0.141	0.044	0.001	15.2%
2001 "All Snags" n=140, chi-sq=10.408, df=1, p=0.001					2001 "Deciduous Snags" n=140, chi-sq=3.262, df=1, p=0.071				
Constant	-3.045	0.449	<0.001		Constant	-2.711	0.433	<0.001	
Density	0.093	0.029	0.001	9.8%	Density	0.111	0.059	0.063	11.7%

Figure 4-7: Secondary cavity-nester (coarse-scale) relative predicted probability of selecting a nest location relative to snag density (≥ 12.5 cm DBH).



4.3.4 Selection of Nest-Snags: Fine Scale

All 3 cavity-nesting guilds used nest-snags with a significantly larger mean DBH than what was available (Table 4-4). WCN's did not use nest-snags with a mean height different than what was available; however, SCN's used nest-snags that were significantly

shorter than those available and PCN's used snags that were significantly taller than available deciduous snags (Table 4-4).

Table 4-4: Difference between nest-snag DBH (cm) and height (m) (including standard error) for snags used by species, grouped into nesting guilds, and snags available within 0.04ha of the nest-snag. Only snags ≥ 14.5 cm DBH were deemed available to PCN's; snags ≥ 12.5 cm DBH were deemed available to WCN's and SCN's. Bold type indicates significance at $\alpha = 0.10$.

Guild ¹	DBH		<i>t</i>	<i>p</i>	Height		<i>t</i>	<i>p</i>
	Nest Snag	Available			Nest Snag	Available		
All Snags								
PCN	34.6 (1.23)	23.4 (0.31)	9.457	<0.001	16.3 (0.82)	15.4 (0.19)	1.170	0.242
WCN	34.3 (1.96)	29.1 (0.96)	-2.042	0.043	10.1 (0.90)	10.4 (0.59)	0.199	0.843
SCN	33.0 (1.30)	25.1 (0.38)	5.793	<0.001	11.0 (1.04)	14.9 (0.26)	-4.070	<0.001
Deciduous Snags								
PCN	34.6 (1.23)	25.7 (0.52)	6.454	<0.001	16.3 (0.82)	14.6 (0.34)	1.890	0.059
WCN	34.3 (1.96)	29.3 (1.06)	-2.002	0.047	10.1 (0.90)	8.5 (0.63)	-1.441	0.157
SCN	33.0 (1.30)	27.3 (0.55)	3.839	<0.001	11.0 (1.04)	13.5 (0.43)	-2.175	0.030

¹ PCN = Primary cavity-nesters; WCN = Weak cavity-nesters; SCN = Secondary cavity-nesters. (see introduction).

Logistic regression identified DBH as a significant variable in predicting PCN nest selection but did not identify snag height as significant (Table 4-5). DBH was also non-linear in the "all snag" (Box-Tidwell test; chi-square = 21.802, df = 1, $p = <0.001$) and "deciduous" models (Box-Tidwell test; chi-square = 16.309, df = 1, $p = <0.001$). Above a snag DBH of 14.5 cm, the relative predicted probability of a PCN selecting a nest-snag increased with DBH, until approximately 45cm, then declined (Figure 4-8a). The models correctly predicted 93.0% and 86.2% of the used and available snags and accounted for 21.9% and 15.5% of the variation between used and available snag DBH, for "all snag" and "deciduous snag" models, respectively.

Table 4-5: Fine-scale logistic regression models describing the importance of snag DBH and snag height in the selection of nest-snags for primary (PCN), weak (WCN), and secondary (SCN) cavity-nesting guilds. Odds ratio indicates the percent change in the probability of nesting associated with a 1% change in the variable.

Variables	B	SE	p	Odds Ratio	Variables	B	SE	p	Odds Ratio
PCN "All Snags" n=910, chi-sq= 83.120, df=2, p=<0.001					PCN "Deciduous Snags" n=463, chi-sq=49.854, df=2, p=<0.001				
Constant	-14.075	2.45	<0.001		Constant	-11.757	2.424	<0.001	
DBH	1.347	0.334	<0.001	84.7%	DBH	1.164	0.328	<0.001	20.3%
(DBH)					(DBH)				
ln(dbh)	-0.277	0.074	<0.001	-24.2%	ln(dbh)	-0.241	0.072	0.001	-21.4%
<u>Variables not in the Model:</u>					<u>Variables not in the Model:</u>				
Snag Ht.	----	----	0.917	----	Snag Ht.	----	----	0.103	----
WCN "All Snags" n=156, chi-sq=24.186, df=4, p=<0.001					WCN "Deciduous Snags" n=122, chi-sq=20.362, df=4, p=<0.001				
Constant	-19.284	5.617	<0.001		Constant	-16.828	5.401	<0.001	
Snag Ht.	2.152	0.695	0.002	59.8%	Snag Ht.	2.191	0.731	0.002	94.7%
(Ht.)					(Ht.)				
ln(ht.)	-0.660	0.210	0.002	-47.4%	ln(ht.)	-0.643	0.215	0.002	-47.4%
DBH	1.469	0.678	0.03	34.6%	DBH	1.164	0.646	0.072	20.2%
(DBH)					DBH				
ln(dbh)	-0.314	0.150	0.036	-27.0%	ln(dbh)	-0.250	0.143	0.08	-22.1%
SCN "All Snags" n=795, chi-sq=52.591, df=2, p=<0.001					SCN "Deciduous Snags" n=400, chi-sq=20.222, df=2, p=<0.001				
Constant	-3.500	0.454	<0.001		Constant	-2.900	0.514	<0.001	
DBH	0.074	0.013	<0.001	7.7%	DBH	0.055	0.014	<0.001	5.6%
Snag Ht.	-0.086	0.018	<0.001	-8.2%	Snag Ht.	-0.045	0.019	0.016	-4.4%

Logistic regression identified DBH and snag height as significant variables in predicting WCN nest selection, in both "all snag" and "deciduous snag" models, in non-linear form ("all snag" Box-Tidwell test: chi-square = 19.735, df = 2, $p = <0.001$; "deciduous" Box-Tidwell test: chi-square = 16.018, df = 2, $p = <0.001$) (Table 4-5). The relative predicted probability of a WCN selecting a nest-snag increased with snag height and DBH until approximately 11 m and 40 cm DBH, then began to decline (Figure 4-8b). The models correctly predicted 86.5% and 82.8% of the used and available snags and accounted for 24.4% and 22.2% of the variation between used and available snags for the "all snag" and "deciduous snag" models, respectively.

DBH and snag height were also significant variables in predicting SCN nest selection for both "all snag" and "deciduous" models (Table 4-5). The relative predicted probability of SCN's selecting a nest-snag generally increased with DBH, however, it decreased with snag height (Figure 4-8c). The models correctly predicted 92.2% and 85.8% of the used and available snags and accounted for 14.1% and 6.6% of the variation between used and available snag attributes for the "all snag" and "deciduous snag" models, respectively.

Figure 4-8: Fine-scale relative predicted probabilities of nest-snag use relative to snag DBH and snag height for primary, weak, and secondary cavity-nesting guilds. Only snag variables that were significant in the logistic regression model were graphed.

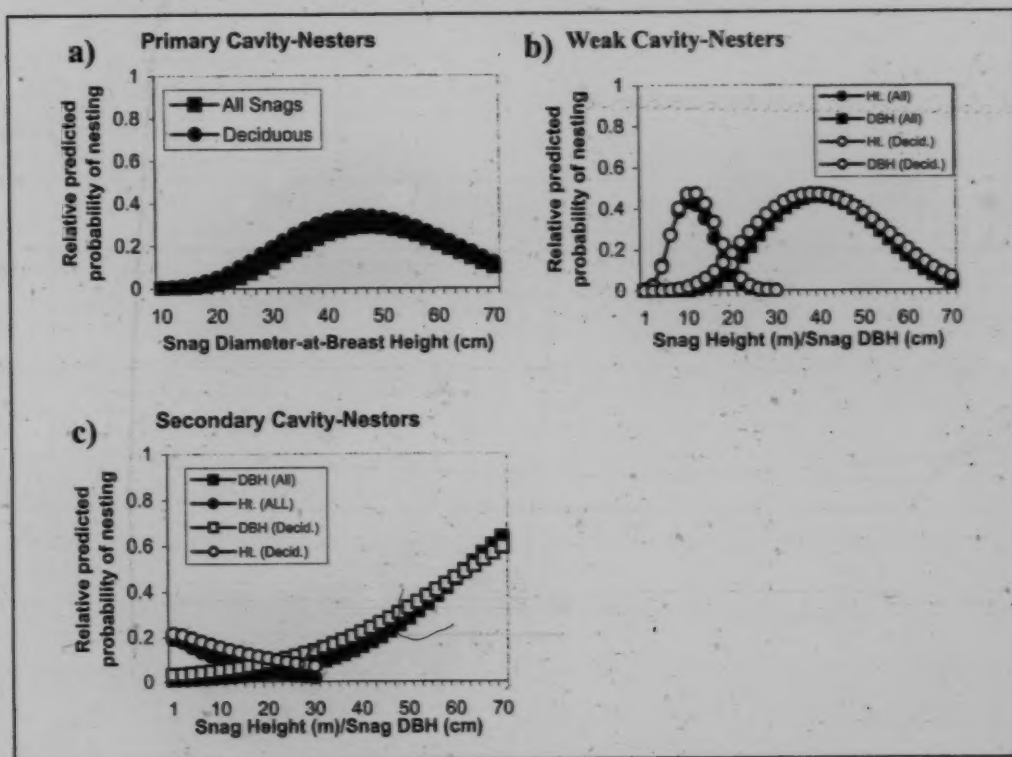


Table 4-6 summarizes guild-level snag attributes gathered for cavity nests located from 1999-2001. Means and confidence intervals are derived from nests within unburned and unharvested, burned and unharvested, and burned and salvage-logged habitat types; therefore, values provided are from a wide spectrum of available habitats.

Table 4-6: Summary of nest-s snag attributes from cavity nests located from 1999-2001 in 3 habitat types: 1) unburned/unharvested, 2) burned/unharvested, and 3) burned/harvested. Stand origins were all originally ~100 year-old boreal mixedwood forest.

Guild	Attribute	Mean	Standard Error	Min.	Max.	95% Confidence Interval
PCN (n=65)	DBH (cm)	34.7	1.220	14.5	54.2	32.2 - 37.1
	Snag Height (m)	16.5	0.825	2.7	26.4	14.8 - 18.1
	Cavity Height (m)	8.3	0.603	2.1	18.4	7.1 - 9.5
	Snag Density (0.04ha)	11.4	1.107	1.0	33.0	9.2 - 13.6
	Deciduous Snag Density (0.04ha)	5.5	0.667	0.0	32.0	4.1 - 6.8
WCN (n=21)	DBH (cm)	34.3	1.956	19.4	53.0	30.2 - 38.4
	Snag Height (m)	10.1	0.899	2.8	18.2	8.2 - 12.0
	Cavity Height (m)	7.9	0.794	1.3	14.6	6.3 - 9.6
	Snag Density (0.04ha)	6.4	1.208	1.0	23.0	3.9 - 9.0
	Deciduous Snag Density (0.04ha)	4.8	0.767	1.0	14.0	3.2 - 6.4
SCN (n=61)	DBH (cm)	33.6	1.366	12.5	61.5	30.9 - 36.3
	Snag Height (m)	11.6	1.078	1.2	30.2	9.5 - 13.8
	Cavity Height (m)	6.3	0.674	0.8	20.2	4.9 - 7.6
	Snag Density (0.04ha)	12.4	1.047	1.0	35.0	10.3 - 14.5
	Deciduous Snag Density (0.04ha)	5.8	0.655	0.0	28.0	4.5 - 7.1

4.4 Discussion

Understanding the dynamics of the cavity-nesting bird community can provide managers with a better understanding of the impacts these species face with increased industrial development. This study addressed general associations among cavity-nesting species at both the species and guild level, across different habitat types. The results also indicate that certain easy to identify, and manageable, forest characteristics are used by cavity-nesting species in selecting nest locations.

4.4.1 Species and Guild Associations Between Cavity Nesters

Composition of the cavity-nesting community in this study was similar to that of Martin and Eadie (1999), and the overall relationships generally support their findings. In both studies, the primary and weak cavity nesting guilds excavated cavities primarily from trembling aspen, and WCN's rarely used existing cavities. However, differences between study systems at the level of the secondary cavity-nesting guild likely reflect differences in the underlying dynamics of the systems. In this study, the natural disturbance (fire) that created the early post-fire habitat resulted in increased use of natural cavities by SCN's, compared to the older-aged and relatively static system Martin and Eadie monitored, where SCN's utilized more cavities created by PCN's.

Post-fire forests are known to attract a pulse of cavity-nesting species immediately following disturbance (Hutto 1995; Villard and Schieck 1997). The period of monitoring in this study (1-3 years post-fire) indicates an initial post-fire lag effect in guild associations. I predicted PCN's and SCN's to be positively and significantly correlated, however this relationship, as demonstrated by abundance and nest correlations, did not emerge until the third year post-fire. *Picoides* woodpeckers (PCN's) were very abundant initially and gradually declined (D. Stepnisky, unpubl. data), while SCN's experienced an increase with time post-fire. WCN's did not show a clear trend, although annual fluctuations in chickadee and nuthatch (WCN) numbers are not uncommon, with irruptions occurring due to weather and food availability (Smith 1991; Semenchuk 1992). Relationships between PCN – WCN and WCN – SCN were also significant by the third year (unlike Martin and Eadie (1999)); however, nest correlations did not corroborate this result. Long-term monitoring is required to determine whether the relative proportions of these guilds will stabilize to a level where relationships remain significant.

Excluding older types of excavations (roosts, foraging, and partial excavations) from the flow of cavity use in this study resulted in a conservative examination of relationships between individual species. Within this system, there was high utilization of naturally occurring cavities. All American Kestrels and European Starlings (n=10), 3 out of 4 duck species, and 1 out of 3 Northern Hawk-Owls (the other 2 were in "broken

top" snags) used natural cavities in burned out trembling aspen trees. Many Tree Swallow and House Wren nests were inside cracks and large fissures of old and decaying snags, indicating that not only wildfire, but also the condition of the forest pre-fire, provided suitable cavities for secondary cavity nesters.

The proportion of PCN cavity use by SCN's was lower than expected, considering the significant correlation between guilds. However, the body sizes of the SCN's in this study do not correspond well with the proportion of available PCN cavities. Pileated woodpecker, capable of supplying a cavity suitable for the largest of SCN's in this study (Bonar 2000), was not an abundant species (2 cavities located). The other SCN's were generally much smaller in body size than most of the available woodpecker cavities, and it has been reported that larger than necessary cavity entrances can increase the risk of predation and competition (Newton 1998). House Wrens, which tended to reuse a number of different PCN cavities, may have addressed this issue by jamming the cavity full of twigs before making a nest among them. Interestingly, 66.6% of Mountain Bluebird nests ($n=3$) were in old Three-toed Woodpecker cavities. Mountain Bluebirds are quite similar in size to Three-toed woodpeckers and their preference to forage and nest near open areas (edges of salvage-logged cutblocks) may account for this close relationship, although sample size was very small. Furthermore, pairing of these species is likely to occur only under certain circumstances (i.e. fire adjacent to natural or harvested openings).

Finally, trembling aspen was identified as the most preferred tree or snag species for cavity excavation, supporting research from both western (Harestad and Keisker 1989; Martin and Eadie 1999) and eastern forests (Runde and Capen 1987). Even WCN's, occurring throughout the study area, preferred to excavate their own nest cavities within decaying aspen snags. WCN's did use 2 PCN cavities for nesting, but did not provide any cavities to SCN's. These results support Martin and Eadie's argument that WCN's should be considered an independent guild and not lumped with primary or secondary cavity-nesters when evaluating the cavity-nesting bird community. While my results are correlative, they identify ecological relationships that can be further examined through hypothesis testing. Recognizing the cavity-nesting bird community lives within

a hierarchal web-like structure allows managers to better predict and forecast the precipitating effects of forest harvesting on these species.

4.4.2 Nest Selection by Cavity-Nesting Guilds

Density of snags was a significant predictor of PCN and SCN nest-locations at what was defined as a coarse-scale in this study (0.04 ha surrounding the nest-snag), or third order selection according to Johnson (1980). Low variation explained by the models however, 5.3% - 41.2% depending on the year, indicate that there are other factors at work besides the availability of snags >12.5 cm DBH. The extremely high and consistent probability of PCN's nesting in locations (0.04 ha) with >25 deciduous snags/0.04 ha reinforces the importance of hardwood trees (primarily aspen) to cavity-excavators. However, variation explained in the deciduous snag models (6.4% - 11.0%) were also low, and thus the results need to be interpreted cautiously.

There is no question that nesting substrate (trees/snags) is an ultimate requirement for cavity-nesting species, but many other factors can influence nest-selection and nesting success. Swallow et al. (1986) and Sedgwick and Knopf (1990) indicated forest characteristics, such as total snag basal area and tree species diversity, were more reliable predictors of bird use than snag characteristics, as these measures may also represent foraging requirements. Foraging resources for cavity-nesting birds must be considered, and not necessarily assumed adequately addressed, if nesting features are preserved in the form of snags (Weikel and Hayes 1999). This study, along with many others, identified deciduous trees as preferred nesting substrate. However, many studies on the foraging ecology of certain PCN species, most of which were abundant within this study, have identified coniferous trees as preferred foraging resources (see Caton 1996; Murphy and Lehnhausen 1998; Kreisel and Stein 1999; Hoyt 2000). Thus, the relatively high diversity of cavity-nesting species within boreal regions may be a result of natural mixedwood forests, that provide both nesting and foraging resources.

Nevertheless, this study supports others that indicate snag density is an important factor when considering cavity-nesting birds (see Dickson et al. 1983; Raphael and White 1984; Zarnowitz and Manuwal 1985), although it does not advocate snag density as the

only predictor of cavity-nester nest site selection. WCN's did not use nest-locations with snag densities different than what was randomly available, and food resources were not included in this analysis. Furthermore, this study only examined variation in snag densities within a home range. At broader spatial extents, I would hypothesize that PCN selection of a "home range" (2nd order selection) would be influenced by the density of coniferous trees/snags for foraging.

Within the selected "nest locations" (4th order selection), all guilds used nest-snags with a larger DBH than what was available. These results indicate that cavity-nesting birds are selecting snags above some minimal diameter, considering the size of interior cavity can influence brood size and fledgling success (Newton 1998). However, the probability of PCN's nesting in a snag began to decrease beyond ~45 cm DBH, also indicating some type of disadvantage to very large diameter snags. Perhaps very large trees retain the strength in their sapwood longer than smaller trees, once they have died, limiting a cavity-nesters ability to excavate (Harestad and Keisker 1989). The majority of trees/snags larger than 45 cm DBH in this study were balsam poplar, and at this size and age, have very hard, thick bark. Furthermore, many species, of all 3 guilds, were observed to nest in "broken top" snags. These are snags that have lost their tops (e.g., due to wind) and are usually dead long before the event of fire. Although rot class was not recorded for nest-snags, it is generally inferred that broken top snags, due to heartwood exposure, decay at a quicker rate than intact snags. Thus, broken top snags are usually easier to excavate and/or increase the likelihood of having naturally occurring crevices and cracks.

The probability of WCN's and SCN's selecting a nest-snag also increased with DBH, however, DBH for these guilds may not be a direct indicator of snag selection. SCN's obviously require existing cavities to nest. Granted snags are large enough to support a nest-cavity, other factors are likely more important with this guild's selection process. SCN's also had a higher probability of nesting in shorter snags; observationally, many were broken tops. The combination of increasing DBH with decreasing height may be an indicator of older trees in a later stage of decay, increasing the probability of an available cavity (natural or previously excavated).

Steeger and Hitchcock (1998) report from southeastern British Columbia, Canada, that Red-breasted Nuthatches (which were the most numerous WCN in this study) used nest trees with significantly more sapwood decay than non-nest trees. Presumably, weak excavators are limited by other snag suitability criteria, such as snag hardness, considering they are relatively small birds and most snags are large enough to be considered available for nesting. Logistic regression indicated snag height was a significant predictor in WCN snag selection, however, the probability of use declined beyond ~11 m. The mean height and DBH of WCN nest-snags in this study also suggest used snags are mostly broken tops (confirmed by observation). Schepps et al. (1999), studying tree hardness relative to nest-site selection of primary cavity nesters in Arizona, USA, noted hardness within a tree declined with increasing height. WCN's consistently nested very close to the top of nest-snags in this study (Table 4-4), which may suggest that the relative stage of decay or hardness of the snag is an important characteristic in WCN nest-snag selection.

4.4.3 Study Limitations

This study focused on simple, easy to identify, snag characteristics when addressing the questions of nest selection. The rationale was to identify, if possible, snag characteristics that increased the probability of the "broad" cavity-nesting bird community to select a nest-location or nest-snag, and that could easily be identified by forest harvesting personnel. The models used to examine the questions, both at the coarse and fine scale, were not comprehensive. Ideally, they would include variables addressing food resources and forest characteristics, as each may contribute to the overall selection process. Nevertheless, at the finest scale (nest-snag), food and forest characteristics are not likely to confound the selection process, as they are presumably standardised to a large degree. However, more specific biological factors such as rot class or evidence of fungal decay may be better predictors of actual snag selection – neither was included in those models.

Pooling data can be influential on the outcome of results (Schooley 1994). By pooling species data into guilds, there is a potential to conceal individual species

selection criteria and mask specific effects (Swallow et al. 1986). The only time this was vividly apparent was in the "deciduous snag" model for PCN snag density. If specific species were of interest, conducting the analyses by individual species would be more appropriate. However, there is an obvious trade-off with sample size relating to the power of the analysis, and for management applications, the general relationships may be of greater interest.

4.5 Conservation Implications

This study corroborates Martin and Eadie's (1999) suggestion that the cavity-nesting bird community functions in a similar structure to a food web. By considering cavity-nesting species in this framework, managers can increase their knowledge of system dynamics related to the forest bird community. The nest web in this study suggests a lag in the relationship between cavity-nesting guilds post-fire, as well as a clear and substantial utilization of natural cavities created by fire. Immediate removal of post-fire forests through salvage operations may have unforeseen effects on the cavity-nesting bird community beyond the first few years post-disturbance. Maintaining post-fire habitats should be considered important for the conservation of cavity-nesting species.

To ensure long-term persistence of the cavity-nesting bird community among managed forests, maintaining nesting requirements is an essential, although not exclusive, consideration. Snags, and live trees in a late state of fungal decay, are critical nesting substrate for cavity-nesting birds. Snags do not typically get harvested because of their economic value (rotten timber), but can impose working hazards for loggers. Today's mechanized harvesting operations alleviate the safety issues around leaving standing snags while working within cutblocks, therefore, every effort should be made to retain snags within cutblock planning. Even after a fire, when snags are present at very high densities, the condition of the forest before the fire is an apparent factor in nest-selection (i.e. use of many broken top snags, broken before the fire). For the benefit of all cavity-nesters, all snags >15 cm DBH should be retained at the highest density possible for both foraging and nesting resources, and deciduous (particularly trembling aspen) snags of all height, larger than 30 cm DBH, are of special nesting importance.

Where possible, snags should be protected by variable retention of live trees for structural support (reduce chances of blow-down) and maintenance of other forest characteristics. This study does not address the issue of live retention within cutblocks, but recognizes the need for some proportion of live trees to be spared from harvesting, as these will eventually become future snags when the cutblock reaches a later seral stage.

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Chapter 5: Thesis Conclusions and Management Implications

5.1 Summary and Conclusions

I examined the response of the forest bird community, 1-3 years following a wildfire, within burned/unharvested (Leave), unburned/unharvested (Control), and burned/salvage-logged (Salvage) areas. Each treatment originated from ≥ 100 year old deciduous-dominated mixedwood boreal forest. Leave and Salvage areas were dynamic, and both bird species richness and abundance generally increased with time post-disturbance. Abundance of these species also tended to increase with the intensity of burned areas. There was little difference between abundance and richness of breeding songbirds among the 3 treatments during the first year. However by the third year post-fire, Leaves supported significantly more species, in greater abundance, than either Controls or Salvages (see also Schulte and Niemi 1998) (Table 5-1). Salvaged areas supported the fewest number of species, although by the third year, abundance was not statistically different from Controls.

Breeding indices also revealed significantly higher values for, or no difference in, relative reproductive activity, for several bird species within Leaves, compared to Controls and Salvages, during the second and third year of monitoring. Early-seral Leave treatments appeared to sustain the quality of breeding habitat for certain ubiquitous species from late-seral unburned Control treatments, which was not evident in the salvage-logged treatments (Table 5-1). Consistent with Hutto (1995) and Imbeau et al. (1999), I found that Leave areas contained many species typically regarded as close associates of post-fire forests, particularly cavity-nesters and flycatchers. Salvaged areas supported relatively few cavity-nesting species due to the removal of trees and snags. However, some species (primarily Northern Flickers) were found nesting within cutblocks when suitable snags were retained. Species similarity within each treatment was distinct, based on Morisita-Horn indices. However, some increase in similarity between Leaves and Controls was apparent 3-years post-fire. This is notably earlier than the ~28 years post-fire convergence of burned and harvested stands suggested by Hobson

and Schieck (1999). Interestingly, this study indicated a decrease in similarity between Salvage and Control bird communities during the initial stages of succession (Table 5-1).

Table 5-1: Summary of thesis results obtained from chapters 2, 3, and 4.

Chapter 2: Community-Level Comparisons

Species Abundance:	Leaves Highest	Salvages Lowest
Species Richness:	Leaves Highest	Salvages Lowest
Species Similarity:	Treatment bird communities different each year (1 - 3 Yrs Post-fire)	
	Controls & Leaves Increased Similarity	Controls & Salvages Decreased Similarity
		Leaves & Salvages Fluctuated

Chapter 3: Comparison of Relative Reproductive Activity

Relative Reproductive Activity:	Black-capped Chickadee	No Difference
	Red-breasted Nuthatch	No Difference
Comparison Between Leaves & Controls	Mourning Warbler	Leaves Highest
	Brown Creeper	Leaves Highest
	Yellow-rumped Warbler	Controls Highest
Comparison Among Leaves, Controls, & Salvages	Lincoln's Sparrow	Salvages Highest
	White-throated Sparrow	Leave & Control Highest

Chapter 4: Cavity-Nesting Species Nest Selection

Coarse-Scale Nest Selection:	Relative predicted probability of a species guild nesting within a "nest location" (0.04 ha)	
	Primary Cavity-Nesters:	Increased with Snag Density
	Weak Cavity-Nesters:	No Difference
	Secondary Cavity-Nesters:	Increased with Snag Density
Fine-Scale Nest Selection:	Relative predicted probability of a species guild selecting a "nest-snag" within a "nest-location" (0.04 ha)	
	Primary Cavity-Nesters:	Increased with snag DBH til ~45cm before decreasing
	Weak Cavity-Nesters:	Increased with snag DBH til ~45cm and snag height till ~10m before decreasing
	Secondary Cavity-Nesters:	Increased with snag DBH and decreased with snag height

This research corroborated others (see Dickson et al. 1983; Raphael and White 1984; Zarnowitz and Manuwal 1985) in identifying a strong relationship between cavity-

nesting birds and increasing snag density, along with a very high preference for trembling aspen as a nest tree. However, many other studies have indicated the preference of conifer trees and snags as foraging resources for certain cavity-nesting species (see Chapter 4); therefore, food requirements for cavity-nesting species should not be assumed adequately addressed by the provision of nest-snags. The relative predicted probability of cavity-nesting birds that excavate their own cavities increased with the diameter of snags, but suggested very large snags (>45 cm DBH), decreased certain cavity-excavators probability of nesting (Table 5-1). The relative hardness of snags was assumed to influence these species, as many nests were located in "broken-top" snags in later stages of decay. Therefore, maintaining the condition of a forest before wildfire may be just as important, to provide additional habitat heterogeneity (e.g. older, broken-top snags), as maintaining forests post-fire.

5.2 Management Implications

This study clearly demonstrates that post-fire forests have substantial ecological value for the forest bird community and that salvage-logging significantly diminished this value. Where the conservation of forest birds is a primary concern, harvesting of forests post-fire is not recommended. However, in forests managed for timber production, the following guidelines are proposed as an attempt to balance economic and socio-political factors.

As a first step, the provincial government and forest companies should recognize the ecological importance of wildfire on the landscape, and the unique habitat it creates, by explicitly considering these values in forest management planning. This requires a fundamental shift in the current approach, which considers harvest as a surrogate to fire in maintaining biotic pattern and process. Where salvage-logging of burned areas is planned, the following should be considered:

Landscape-Level

(1) Leave representative proportions of older-aged post-fire forests throughout Forest Management Areas (FMA's). The question of what amount of burned forest to maintain is of utmost concern, but has not been addressed (see research recommendations, below).

This study identified areas of medium to high intensity burn to be most beneficial to those species utilizing burned areas. Maintaining representative proportions of these habitats across the landscape, as is currently done for other habitats of concern, may be an initial starting point (e.g., considering "older-aged" burned forest a unique stand-type (seral-stage) worth maintaining through forest management planning).

Stand-Level

- (1) Within harvested areas, leave patches of "older-aged" burned forest. This study found that 10 ha patches were valuable to bird species, however, larger patches may support many other species with larger area requirements, for longer periods of time.
- (2) Where possible, patches of burned forest should be adjacent to unburned forest, non-commercial forest and/or riparian buffers. Species movement between habitat types can be impaired by large forest gaps (Machtans et al. 1996; Desrochers and Hannon 1997) and seasonal differences in use, between burned and unburned forest by resident species, has been reported (Kreisel and Stein 1999; Stambaugh and Schmiegelow *in prep.*).
- (3) Within cutblocks, all snags >15 cm DBH (foraging substrate) and deciduous snags >30 cm DBH (nesting substrate) of reduced economic value (later stages of decay), should be retained in the highest density possible to support cavity-nesting birds.

Implementation of the suggested guidelines will require support by the general public, and a strong commitment by the Alberta government and forest companies. Education regarding the role wildfire plays within the boreal forest needs to reach schools, communities and politicians. Alberta's true commitment to sustainable forest management (Alberta Sustainable Resource Development 1999) may depend on a change in perceptions at many levels.

5.3 Future Research

There are still many unknowns regarding the role of fire, and associated effects of salvage-logging, for wildlife. Here, I identify what I consider to be the 2 most important questions for management of boreal forests, relative to this issue:

- (1) How much burned forest of commercial value (i.e. older forests) is required to maintain forest bird populations, specifically those considered post-fire specialists? Addressing this question entails a large, landscape-level experiment, supplemented by simulation modeling. In particular, the identification of potential habitat thresholds, below which populations decline, and probability of persistence is reduced, should be a priority.
- (2) Does burned forest not of commercial value (i.e. younger forest and that comprised of non-commercial tree species) provide habitat opportunities for forest bird species, particularly post-fire specialists? If so, can maintenance of this habitat off-set the harvesting of "older-aged" burned forest? The majority of forest within a fire perimeter often remains unharvested, however, unharvested forest types are not likely to provide the same ecological values as older forests. For example, cavity-nesting birds require larger diameter snags to build nest cavities and younger forests do not typically contain these resources. However, how significant are these habitats for foraging of cavity-nesting species and the general utilization of non-cavity nesting species?

While this is far from a comprehensive list of research questions, the ones outlined are of significant ecological and management interest, and provide substantial scope for future research in this area.

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Appendix 1: Classification system used for describing level of burn intensity within Leave treatments (burned & unharvested) for the Chip Lake wildfire (from Hutto 1995).

Classification	Description of Burn Intensity
0	Unburned
1	More than 60% of trees having green leaves of needles
2	40% - 60% of trees having green leaves of needles
3	5% - 40% of trees having green leaves of needles
4	<5% of trees having green leaves or needles
5	No trees with green leaves or needles
6	Mostly broken stumps with few standing trees

Appendix 2: All species documented during point count surveys in 1999, 2000, and 2001 within the Chip Lake study area.

	Species Code	Species	Scientific Name
1	ALFL	Alder Flycatcher	<i>Empidonax alnorum</i>
2	AMGO	American Goldfinch	<i>Carduelis tristis</i>
3	AMKE	American Kestrel	<i>Falco sparverius</i>
4	AMRE	American Redstart	<i>Setophaga ruticilla</i>
5	AMRO	American Robin	<i>Turdus migratorius</i>
6	BAWW	Black-and-white Warbler	<i>Mniotilta varia</i>
7	BBWO	Black-backed Woodpecker	<i>Picoides arcticus</i>
8	BCCH	Black-capped Chickadee	<i>Parus atricapillus</i>
9	BGNW	Black-throated Green Warbler	<i>Dendroica virens</i>
10	BHCO	Brown-headed Cowbird	<i>Molothrus ater</i>
11	BHVI	Blue-headed Vireo	<i>Vireo solitarius</i>
12	BOCH	Boreal Chickadee	<i>Parus hudsonicus</i>
13	BRBL	Brewer's Blackbird	<i>Euphagus cyanocephalus</i>
14	BRCR	Brown Creeper	<i>Certhis americana</i>
15	BWHA	Broad-winged Hawk	<i>Buteo platypterus</i>
16	CCSP	Clay-coloured Sparrow	<i>Spizella pallida</i>
17	CEWX	Cedar Waxwing	<i>Bombacilla cedrorum</i>
18	CHSP	Chipping Sparrow	<i>Spizella passerina</i>
19	COSN	Common Snipe	<i>Gallinago gallinago</i>
20	COWA	Connecticut Warbler	<i>Oporornis agilis</i>
21	COYE	Common Yellowthroat	<i>Geothlypis trichas</i>
22	DEJU	Dark-eyed Junco	<i>Junco hyemalis</i>
23	DOWO	Downy Woodpecker	<i>Picoides pubescens</i>
24	EUST	European Starling	<i>Sturnus vulgaris</i>
25	GCKI	Golden-crowned Kinglet	<i>Regulus satrapa</i>
26	GGOW	Great-gray Owl	<i>Strix nebulosa</i>
27	GRJA	Gray Jay	<i>Perisoreus canadensis</i>

Appendix 2 (continued): All species documented during point count surveys in 1999, 2000, and 2001 within the Chip Lake study area.

	Species Code	Species	Scientific Name
28	HAWO	Hairy Woodpecker	<i>Picoides villosus</i>
29	HETH	Hermit Thrush	<i>Catharus guttatus</i>
30	HOWR	House Wren	<i>Troglodytes aedon</i>
31	KILL	Killdeer	<i>Charadrius vociferus</i>
32	LCSP	Le Conte's Sparrow	<i>Ammodramus leconteii</i>
33	LEFL	Least Flycatcher	<i>Empidonax minimus</i>
34	LISP	Lincoln's Sparrow	<i>Melospiza lincolni</i>
35	MOBL	Mountain Bluebird	<i>Sialia currucoides</i>
36	MOWA	Mourning Warbler	<i>Oporornis philadelphia</i>
37	NHOW	Northern-hawk Owl	<i>Surnia ulula</i>
38	NOFL	Northern Flicker	<i>Colaptes auratus</i>
39	NSWO	Northern Saw-whet Owl	<i>Aegolius acadicus</i>
40	OSFL	Olive-sided Flycatcher	<i>Contopus borealis</i>
41	OVEN	Ovenbird	<i>Seiurus aurocapillus</i>
42	PHVI	Philadelphia Vireo	<i>Vireo philadelphicus</i>
43	PISI	Pine Siskin	<i>Carduelis pinus</i>
44	PIWO	Pileated Woodpecker	<i>Dryocopus pileatus</i>
45	PUFI	Purple Finch	<i>Carpodacus purpureus</i>
46	RBGR	Rose-breasted Grosbeak	<i>Pheucticus ludovicianus</i>
47	RBNU	Red-breasted Nuthatch	<i>Sitta canadensis</i>
48	RCKI	Ruby-crowned Kinglet	<i>Regulus calendula</i>
49	REVI	Red-eyed Vireo	<i>Vireo olivaceus</i>
50	RTHA	Red-tailed Hawk	<i>Buteo jamaicensis</i>
51	RUGR	Ruffed Grouse	<i>Bonasa umbellus</i>
52	RWBL	Red-winged Blackbird	<i>Agelaius phoeniceus</i>
53	SOSA	Solitary Sandpiper	<i>Tringa solitaria</i>
54	SOSP	Song Sparrow	<i>Melospiza melodia</i>
55	SWTH	Swainson's Thrush	<i>Catharus swainsoni</i>
56	TEWA	Tennessee Warbler	<i>Vermivora peregrina</i>
57	TRSW	Tree Swallow	<i>Tachycineta bicolor</i>
58	TTWO	Three-toed Woodpecker	<i>Picoides tridactylus</i>
59	VESP	Vesper Sparrow	<i>Poocetes gramineus</i>
60	WAVI	Warbling Vireo	<i>Vireo gilvus</i>
61	WBNU	White-breasted Nuthatch	<i>Sitta carolinensis</i>
62	WETA	Western Tanager	<i>Piranga ludoviciana</i>
63	WIWR	Winter Wren	<i>Troglodytes troglodytes</i>
64	WTSP	White-throated Sparrow	<i>Zonotrichia albicollis</i>
65	WWCR	White-winged Crossbill	<i>Loxia leucoptera</i>
66	WWPE	Western Wood-Pewee	<i>Contopus sordidulus</i>
67	YBSA	Yellow-bellied Sapsucker	<i>Sphyrapicus varius</i>
68	YRWA	Yellow-rumped Warbler	<i>Dendroica coronata</i>
69	YWAR	Yellow Warbler	<i>Dendroica petechia</i>

Appendix 3: Control treatment species list indicating mean species abundance among all 3 years (1999 - 2001) in decreasing order, including percentage of community composition.

	Species	Year Present	Spp. Code	Nest Site ¹	Diet ²	Forage Site ³	Abund. Index	% of Community
1	White-throated Sparrow	1,2,3	WTSP	G	I	G	0.607	14.04
2	Yellow-rumped Warbler	1,2,3	YRWA	T	I	F	0.435	10.07
3	Swainson's Thrush	1,2,3	SWTH	S	I	F	0.359	8.30
4	Red-breasted Nuthatch	1,2,3	RBNU	C	I	B	0.337	7.79
5	Ovenbird	1,2,3	OVEN	G	I	G	0.333	7.71
6	Mourning Warbler	1,2,3	MOWA	G-↓S	I	F-G	0.323	7.47
7	Tennessee Warbler	1,2,3	TEWA	G	I	F	0.289	6.68
8	Golden-crowned Kinglet	1,2,3	GCKI	T	I	F	0.206	4.76
9	Red-eyed Vireo	1,2,3	REVI	S-T	I	F	0.139	3.22
10	Black-throated Green Warb.	1,2,3	BGNW	T	I	F	0.133	3.07
11	Least Flycatcher	1,2,3	LEFL	T	I	A	0.124	2.87
12	Chipping Sparrow	1,2,3	CHSP	↓T	I	G-F	0.112	2.60
13	American Redstart	1,2,3	AMRE	↓T-S	I	F	0.102	2.36
14	Rose-breasted Grosbeak	1,2,3	RBGR	T-S	I-S	F	0.088	2.04
15	Black-capped Chickadee	1,2,3	BCCH	C	I-S	F-B	0.083	1.93
16	Blue-headed Vireo	1,2,3	BHVI	T	I	F	0.075	1.73
17	Warbling Vireo	1,2,3	WAVI	T	I	F	0.071	1.65
18	Pine Siskin	1,2,3	PISI	T	S-I	F-G	0.060	1.38
19	Brown Creeper	2,3	BRCR	Cr.	I	B	0.053	1.22
20	Hermit Thrush	1,2,3	HETH	G-↓T	I	G-F	0.048	1.10
21	Yellow-bellied Sapsucker	1,2,3	YBSA	C	I	B	0.048	1.10
22	Lincoln's Sparrow	1,2,3	LISP	G	I	G	0.041	0.94
23	Yellow Warbler	1,2,3	YWAR	S	I	F	0.037	0.86
24	Gray Jay	2,3	GRJA	T	O	G-F	0.036	0.82
25	Western Tanager	1,2,3	WETA	T	I	F	0.024	0.55
26	Philadelphia Vireo	2	PHVI	T	I	F	0.020	0.47
27	Boreal Chickadee	1,2,3	BOCH	C	J	B	0.019	0.43
28	White-winged Crossbill	3	WWCR	T	S	F-G	0.019	0.43
29	Hairy Woodpecker	2,3	HAWO	C	I	B	0.015	0.35
30	Brown-headed Cowbird	1,2,3	BHCO	Par.	I	G	0.014	0.32
31	Dark-eyed Junco	3	DEJU	G	S-I	G	0.010	0.24
32	Cedar Waxwing	2	CEWX	T	Fr	F	0.007	0.15
33	Connecticut Warbler	1	COWA	G-S	I	G-F	0.007	0.15
34	Three-toed Woodpecker	2	TTWO	C	I	B	0.007	0.15
35	White-breasted Nuthatch	1	WBNU	C	I	B	0.007	0.15
36	Broad-winged Hawk	1,2	BWHA	T	V	H	0.005	0.12
37	American Goldfinch	2	AMGO	S-↓T	S	F-G	0.003	0.08
38	American Robin	1,3	AMRO	T	I	G	0.003	0.08
39	Black-and-white Warbler	3	BAWW	G	I	B	0.003	0.08
40	Common Yellowthroat	3	COYE	↓S	I	F	0.003	0.08
41	Downy Woodpecker	2,3	DOWO	C	I	B	0.003	0.08
42	Northern Saw-whet Owl	2	NSWO	C	V	H	0.003	0.08
43	Purple Finch	3	PUFI	T	S-I	G-F	0.003	0.08
44	Ruby-crowned Kinglet	3	RCKI	T	I	F	0.003	0.08
45	Western Wood-Pewee	1	WWPE	T	I	A	0.003	0.08
46	Pileated Woodpecker	2	PIWO	C	I	B	0.002	0.04
47	Red-tailed Hawk	1	RTHA	T	V	H	0.002	0.04

¹ G = Ground, S = Shrub, T = Tree, C = Cavity, Cr. = Crevice, Par. = Parasite, ↓ = Low

² I = Insects, S = Seeds, V = Vertebrates, O = Omnivore, Fr. = Fruit, Aq. = Aquatic Insects

³ G = Ground, F = Foliage Gleaner, A = Aerial, B = Bark Gleaner, H = Hunts

^{1,2,3} From Ehrlich et al. 1988 (* only dominant behaviours listed)

Appendix 4: Leave treatment species list indicating mean species abundance among all 3 years (1999 - 2001) in decreasing order, including percentage of community composition.

	Species	Year Present	Spp. Code	Nest Site ¹	Diet ²	Forage Site ³	Abund. Index	% of Community
1	White-throated Sparrow	1,2,3	WTSP	G	I	G	0.956	18.26
2	Mourning Warbler	1,2,3	MOWA	G-↓S	I	F-G	0.633	12.09
3	Lincoln's Sparrow	1,2,3	LISP	G	I	G	0.388	7.41
4	Common Yellowthroat	1,2,3	COYE	↓S	I	F	0.308	5.88
5	House Wren	1,2,3	HOWR	C	I	G-F	0.291	5.55
6	Red-breasted Nuthatch	1,2,3	RBNU	C	I	B	0.269	5.13
7	Yellow-rumped Warbler	1,2,3	YRWA	T	I	F	0.209	4.00
8	Western Wood-Pewee	1,2,3	WWPE	T	I	A	0.180	3.45
9	Brown Creeper	2,3	BRCR	Cr.	I	B	0.156	2.99
10	Least Flycatcher	1,2,3	LEFL	T	I	A	0.155	2.96
11	Tennessee Warbler	1,2,3	TEWA	G	I	F	0.143	2.73
12	Alder Flycatcher	1,2,3	ALFL	S	I	A	0.138	2.63
13	Chipping Sparrow	1,2,3	CHSP	↓T	I	G-F	0.134	2.57
14	Pine Siskin	1,2,3	PISI	T	S-I	F-G	0.116	2.21
15	Swainson's Thrush	1,2,3	SWTH	S	I	F	0.107	2.04
16	American Robin	1,2,3	AMRO	T	I	G	0.104	1.98
17	Golden-crowned Kinglet	1,2,3	GCKI	T	I	F	0.071	1.36
18	Black-capped Chickadee	1,2,3	BCCH	C	I-S	F-B	0.065	1.24
19	Dark-eyed Junco	1,2	DEJU	G	S-I	G	0.061	1.17
20	Western Tanager	1,2,3	WETA	T	I	F	0.060	1.14
21	Northern Flicker	1,2,3	NOFL	C	I	G	0.054	1.04
22	Ruby-crowned Kinglet	2,3	RCKI	T	I	F	0.053	1.01
23	Blue-headed Vireo	2,3	BHVI	T	I	F	0.049	0.94
24	Three-toed Woodpecker	1,2	TTWO	C	I	B	0.043	0.82
25	Rose-breasted Grosbeak	1,2,3	RBGR	T-S	I-S	F	0.041	0.78
26	Le Conte's Sparrow	1,2	LCSP	G	I-S	G	0.034	0.65
27	Yellow Warbler	2,3	YWAR	S	I	F	0.034	0.65
28	American Redstart	1,2	AMRE	↓T-S	I	F	0.031	0.59
29	Gray Jay	1,2,3	GRJA	T	O	G-F	0.031	0.59
30	White-winged Crossbill	3	WWCR	T	S	F-G	0.031	0.59
31	Ruffed Grouse	1,2,3	RUGR	G	O	G-F	0.027	0.52
32	Warbling Vireo	2,3	WAVI	T	I	F	0.027	0.52
33	Hairy Woodpecker	1,2,3	HAWO	C	I	B	0.026	0.49
34	Ovenbird	1,2,3	OVEN	G	I	G	0.024	0.45
35	Yellow-bellied Sapsucker	1,2,3	YBSA	C	I	B	0.022	0.42
36	European Starling	1,2	EUST	C	I	G-F	0.020	0.39
37	Red-winged Blackbird	1,3	RWBL	reeds	I	G-F	0.017	0.32
38	Brown-headed Cowbird	1,2,3	BHCO	Par.	I	G	0.014	0.26
39	American Kestrel	2,3	AMKE	C	V	H	0.012	0.23
40	Boreal Chickadee	2,3	BOCH	C	I-S	F-B	0.010	0.20
41	Olive-sided Flycatcher	2	OSFL	T	I	A	0.010	0.20
42	Philadelphia Vireo	2,3	PHVI	T	I	F	0.010	0.20
43	Red-eyed Vireo	1,3	REVI	S-T	I	F	0.010	0.20

Appendix 4 (Continued): Leave treatment species list indicating mean species abundance among all 3 years (1999 - 2001) in decreasing order, including percentage of community composition.

Species	Year Present	Spp. Code	Nest Site ¹	Diet ²	Forage Site ³	Abund. Index	% of Community
44 Winter Wren	2	WIWR	C	I	G-F	0.010	0.20
45 Black-backed Woodpecker	1,2	BBWO	C	I	B	0.009	0.17
46 Connecticut Warbler	3	COWA	G-S	I	G-F	0.007	0.13
47 Hermit Thrush	1,3	HETH	G-↓T	I	G-F	0.007	0.13
48 Northern-hawk Owl	2	NHOW	C	V	H	0.007	0.13
49 Purple Finch	3	PUFI	T	S-I	G-F	0.007	0.13
50 Cedar Waxwing	2,3	CEWX	T	Fr	F	0.003	0.06
51 Clay-coloured Sparrow	3	CCSP	G-↓S	I	G	0.003	0.06
52 Red-tailed Hawk	2	RTHA	T	V	H	0.003	0.06
53 American Goldfinch	3	AMGO	S-↓T	S	F-G	0.002	0.03
54 Downy Woodpecker	2	DOWO	C	I	B	0.002	0.03
55 White-breasted Nuthatch	3	WBNU	C	I	B	0.002	0.03

¹ G = Ground, S = Shrub, T = Tree, C = Cavity, Cr. = Crevice, Par. = Parasite, ↓ = Low

² I = Insects, S = Seeds, V = Vertebrates, O = Omnivore, Fr. = Fruit, Aq. = Aquatic Insects

³ G = Ground, F = Foliage Gleaner, A = Aerial, B = Bark Gleaner, H = Hunts

^{1,2,3} From Ehrlich et al. 1988 (* only dominant behaviours listed)

Appendix 5: Salvage treatment species list indicating mean species abundance among all 3 years (1999 - 2001) in decreasing order, including percentage of community composition.

	Species	Year Present	Spp. Code	Nest Site ¹	Diet ²	Forage Site ³	Abund. Index	% of Community
1	Lincoln's Sparrow	1,2,3	LISP	G	I	G	0.901	28.45
2	White-throated Sparrow	1,2,3	WTSP	G	I	G	0.527	16.65
3	Le Conte's Sparrow	1,2,3	LCSP	G	I-S	G	0.423	13.36
4	Dark-eyed Junco	1,2,3	DEJU	G	S-I	G	0.199	6.28
5	House Wren	1,2,3	HOWR	C	I	G-F	0.179	5.64
6	Common Yellowthroat	2,3	COYE	↓S	I	F	0.122	3.86
7	Clay-coloured Sparrow	1,2,3	CCSP	G-↓S	I	G	0.119	3.76
8	Common Snipe	1,2,3	COSN	G	I-Aq	P	0.109	3.44
9	Chipping Sparrow	1,2,3	CHSP	↓T	I	G-F	0.082	2.58
10	Alder Flycatcher	1,3	ALFL	S	I	A	0.077	2.42
11	American Robin	1,2,3	AMRO	T	I	G	0.070	2.20
12	Pine Siskin	1,2,3	PISI	T	S-I	F-G	0.065	2.04
13	Song Sparrow	2,3	SOSP	G-↓S	I-S	G-F	0.034	1.07
14	Mourning Warbler	1,2,3	MOWA	G-↓S	I	F-G	0.024	0.75
15	Brewer's Blackbird	2,3	BRBL	T-G	I	G	0.020	0.64
16	Tennessee Warbler	1,2,3	TEWA	G	I	F	0.020	0.64
17	Western Wood-Pewee	1,2	WWPE	T	I	A	0.017	0.54
18	Cedar Waxwing	2	CEWX	T	Fr	F	0.015	0.48
19	Mountain Bluebird	2,3	MOBL	C	I	A	0.015	0.48
20	Black-capped Chickadee	2,3	BCCH	C	I-S	F-B	0.014	0.43
21	Hermit Thrush	1,2	HETH	G-↓T	I	G-F	0.014	0.43
22	Yellow-rumped Warbler	1,2	YRWA	T	I	F	0.014	0.43
23	Solitary Sandpiper	1,2,3	SOSA	T	I-Aq.	P	0.012	0.38
24	Gray Jay	1,2,3	GRJA	T	O	G-F	0.010	0.33
25	Least Flycatcher	1,2	LEFL	T	I	A	0.010	0.33
26	Red-breasted Nuthatch	2,3	RBNU	C	I	B	0.010	0.33
27	Swainson's Thrush	1	SWTH	S	I	F	0.010	0.33
28	Yellow-bellied Sapsucker	1,2,3	YBSA	C	I	B	0.010	0.33
29	Northern Flicker	1,2	NOFL	C	I	G	0.007	0.21
30	Olive-sided Flycatcher	1	OSFL	T	I	A	0.007	0.21
31	Tree Swallow	2,3	TRSW	C	I	A	0.007	0.21
32	Vesper Sparrow	2	VESP	G	I-S	G	0.007	0.21
33	Red-winged Blackbird	1,2	RWBL	reeds	I	G-F	0.005	0.16
34	American Kestrel	2,3	AMKE	C	V	H	0.003	0.11
35	Ruby-crowned Kinglet	3	RCKI	T	I	F	0.003	0.11
36	American Goldfinch	3	AMGO	S-↓T	S	F-G	0.002	0.05
37	Great-gray Owl	2	GGOW	T	V	H	0.002	0.05
38	Killdeer	2	KILL	G	I	G	0.002	0.05
39	Three-toed Woodpecker	1	TTWO	C	I	B	0.002	0.05

¹ G = Ground, S = Shrub, T = Tree, C = Cavity, Cr. = Crevice, Par. = Parasite, ↓ = Low

² I = Insects, S = Seeds, V = Vertebrates, O = Omnivore, Fr. = Fruit, Aq. = Aquatic Insects

³ G = Ground, F = Foliage Gleaner, A = Aerial, B = Bark Gleaner, H = Hunts

^{1,2,3} From Ehrlich et al. 1988 (* only dominant behaviours listed)

Appendix 6: Mean weights (g) for terrestrial invertebrate sweep net collections, taken within each treatment at Chip Lake during the 3rd year (2001) post-fire and harvest.

Classification	Common Name	Collection 1 (Jun15-Jun21)			Collection 2 (July06-July09)			Mean		
		C	L	S	C	L	S	C	L	S
Class Gastropoda	snails, slugs				0.001			0.001		
Class Archnida										
Order Araneae	spiders	0.013	0.031	0.003	0.016	0.022	0.007	0.014	0.027	0.005
Order Acarnia	miles, ticks	0.001	0.002	0.001	0.001	0.002		0.001	0.002	0.001
Class Hexapoda										
Order Diptera	flies									
<i>Nematocera</i>	mosquito, gnat	0.005	0.005	0.002	0.008	0.009	0.001	0.007	0.007	0.001
<i>Brachycera</i>	horseflies, robberflies	0.028	0.246	0.017	0.023	0.029	0.244	0.026	0.137	0.131
<i>Cyclorrhapha</i>	houseflies, blowflies	0.039	0.031	0.008	0.530	0.049	0.017	0.284	0.040	0.013
Order Homoptera										
<i>Auchenorrhyncha</i>	leafhoppers	0.007	0.384	0.043	0.010	0.087	0.206	0.008	0.236	0.125
<i>Stenorrhyncha</i>	plant lice	0.002	0.004	0.001	0.001	0.009	0.004	0.002	0.006	0.003
	aphids	0.001	0.001	0.000	0.006	0.000	0.006	0.003	0.001	0.003
Order Hymenoptera	bee, wasp									
<i>Symphyta</i>	sawfly, horntail	0.002	0.003	0.005	0.004	0.001	0.009	0.003	0.002	0.007
<i>Apocrita</i>	parasitic wasps	0.008	0.028	0.002	0.015	0.012	0.013	0.011	0.020	0.007
	Family Formicidae ant	0.002	0.001	0.001	0.001	0.001	0.001	0.001	0.001	0.001
Order Odonata										
<i>Zygoptera</i>	damsel flies	0.002						0.001		
<i>Anisoptera</i>	dragon flies									
Order Lepidoptera	butterflies, moths		0.004		0.006	0.009	0.007	0.003	0.006	0.003
	larvae	0.330	0.225	0.019	0.031	0.018	0.015	0.181	0.121	0.017
Order Collembolla	springtails									
Order Psocoptera	bark lice		0.002		0.001	0.001	0.001		0.001	0.001
Order Thysanoptera	thrips						0.001			0.000
Order Hemiptera	true bugs	0.001	0.006	0.003	0.007	0.011	0.012	0.004	0.009	0.007
Order Coleoptera	beetles	0.014	0.010	0.365	0.005	0.007	0.019	0.009	0.008	0.192
	Family Carabidae		0.004			0.014	0.008		0.009	0.004
	Family Buprestidae					0.100			0.050	

C=Control: unburned-unharvested; L=Leave: burned-unharvested; S=Salvage: burned-harvested

Appendix 7: Summary of understory vegetation attributes for Controls (unburned/unharvested), Leaves (burned/unharvested), and Salvages (burned/harvested) from 1999 (1st year post-fire) to 2001 (3rd year post-fire).

Year	TRMT	Grass						Short Shrubs						Tall Shrubs					
		Mean % Cov. ¹	Max. %	Min. %	Mean Ht. ²	Max.	Min.	Mean % Cov. ¹	Max. %	Min. %	Mean Ht. ²	Max.	Min.	Mean % Cov. ¹	Max. %	Min. %	Mean Ht. ²	Max.	Min.
1999	Control	n/a	n/a	n/a	n/a	n/a	n/a	36.4 (19.7)	69.6	11.3	0.6 (0.2)	0.9	0.3	17.6 (14.0)	37.5	4.2	3.2 (0.7)	4.6	2.3
	Leaves	n/a	n/a	n/a	n/a	n/a	n/a	30.4 (8.7)	47.1	20.0	0.5 (0.2)	0.8	0.3	6.1 (4.6)	12.5	0.4	1.7 (0.7)	2.9	0.8
	Salvage	n/a	n/a	n/a	n/a	n/a	n/a	5.1 (2.9)	10.4	1.7	0.2 (0.1)	0.3	0.1	1.4 (0.9)	2.5	0.4	0.6 (0.2)	0.9	0.3
2000	Control	15.2 (12.8)	37.7	2.3	0.7 (0.2)	1.1	0.5	38.9 (22.6)	75.2	16.5	0.8 (0.2)	1.1	0.5	12.0 (9.5)	28.4	3.9	3.0 (0.8)	4.2	2.2
	Leaves	35.0 (15.5)	53.3	11.5	0.8 (0.2)	1.0	0.4	30.8 (9.0)	49.0	23.1	0.7 (0.2)	1.0	0.5	5.9 (3.8)	13.2	2.7	1.6 (0.5)	2.6	1.0
	Salvage	32.3 (14.6)	58.8	18.2	0.5 (0.1)	0.7	0.4	7.8 (3.3)	11.3	2.1	0.2 (0.1)	0.4	0.2	3.6 (2.5)	7.4	0.7	0.7 (0.2)	1.0	0.5
2001	Control	23.1 (14.5)	49.0	7.5	0.6 (0.2)	1.0	0.3	39.7 (20.6)	61.3	14.0	0.7 (0.3)	1.1	0.5	16.9 (8.4)	27.7	7.7	1.9 (0.8)	3.3	0.9
	Leaves	42.6 (15.7)	59.2	11.3	0.9 (0.2)	1.1	0.4	38.9 (8.9)	54.2	27.9	0.7 (0.2)	1.1	0.5	10.4 (7.0)	24.0	4.4	1.1 (0.7)	2.4	0.4
	Salvage	49.2 (18.0)	74.4	18.8	0.6 (0.1)	0.8	0.5	21.4 (9.0)	32.5	5.4	0.2 (0.1)	0.2	0.1	7.3 (3.9)	12.9	3.9	0.8 (0.3)	1.1	0.4

¹ Percentage of measurement covering the forest floor, visually obtained from 0.008ha plots (n = 42) averaged per treatment (including SD).

² Mean height of measurement visually obtained from 0.008ha plots (n = 42) and averaged per treatment (including SD).

Appendix 8: Summary of the number of trees of all species, separated into diameter-at-breast-height (DBH) categories, for Controls (unburned/unharvested, Leaves (burned/unharvested), and Salvages (burned/harvested) from 1999 (1st year post-fire) to 2001 (3rd year post-fire).

Year	TRMT	< 2.5cm DBH ¹			2.5-8cm DBH ¹			8-15cm DBH ²			15-23cm DBH ²			23-38cm DBH ²			> 38cm DBH ²		
		Mean # 0.008ha	Max.	Min.	Mean # 0.008ha	Max.	Min.	Mean # (0.04ha)	Max.	Min.	Mean # (0.04ha)	Max.	Min.	Mean # (0.04ha)	Max.	Min.	Mean # (0.04ha)	Max.	Min.
1999	Control	7.5 (4.3)	14	2	3.3 (2.4)	8	1	9.7 (6.5)	24	4	8.1 (3.8)	12	2	3.7 (2.1)	7	2	1.0 (0.6)	2	0
	Leaves	58.1 (35.0)	108	11	1.2 (1.5)	4	0	3.9 (3.3)	9	0	2.3 (2.1)	5	0	2.8 (2.9)	8	0	0.5 (0.7)	2	0
	Salvage	94.8 (65.9)	198	30	0.2 (0.6)	2	0	0.4 (0.6)	1	0	0.0 (0.1)	0	0	0.0	0	0	0.0	0	0
2000	Control	7.5 (5.8)	18	2	2.9 (1.3)	5	1	9.8 (5.8)	21	3	7.6 (3.7)	12	1	4.4 (2.8)	8	1	1.0 (0.5)	2	0
	Leaves	57.1 (35.8)	100	9	1.1 (1.5)	4	0	3.6 (3.4)	9	0	2.4 (2.1)	6	0	2.4 (2.4)	6	0	0.5 (0.5)	1	0
	Salvage	92.5 (66.6)	216	37	0.1 (0.2)	0	0	0.4 (0.4)	1	0	0.1 (0.2)	1	0	0.0	0	0	0.0	0	0
2001	Control	7.0 (6.2)	19	2	2.2 (1.2)	4	1	9.0 (4.8)	18	4	8.6 (4.1)	14	1	5.5 (3.4)	11	2	1.7 (0.8)	3	1
	Leaves	57.1 (35.7)	105	7	2.2 (2.0)	6	0	2.9 (2.5)	8	0	2.1 (2.1)	6	0	2.7 (2.7)	7	0	0.7 (1.0)	3	0
	Salvage	115.0 (56.0)	219	66	0.5 (0.7)	2	0	0.6 (0.8)	2	0	0.0 (0.1)	0	0	0.0	0	0	0.0	0	0

¹ Values are mean number of stems per 0.008ha plot (including Standard Deviation) (n = 42).

² Values are mean number of stems per 0.04ha plot (including Standard Deviation) (n = 42).

Appendix 9: Summary of number of snags and coarse-woody debris within Controls (unburned/unharvested), Leaves (burned/unharvested), and Salvages (burned/harvested) from 1999 (1st year post-fire) to 2001 (3rd year post-fire).

Year	TRMT	Snags (> 12cm DBH)			Coarse Woody Debris (CWD) ¹			
		Mean # (0.04ha)	Mean DBH/ Snag (cm)	Mean Ht./ Snag (m)	Mean # ² (0.04ha)	Mean ³ Diam. (cm)	Mean Length (m)	Mean ⁴ Rot Class
1999	Control	4.0	30.6	7.2	n/a	n/a	n/a	n/a
	Leave	9.7	27.0	12.4	n/a	n/a	n/a	n/a
	Salvage	0.6	12.8	6.2	n/a	n/a	n/a	n/a
2000	Control	3.9	28.9	7.9	4.7	19.9	8.6	4.4
	Leave	11.8	27.1	13.4	5.2	19.7	8.8	3.9
	Salvage	0.6	13.5	5.1	4.5	14.9	3.0	2.5
2001	Control	4.0	32.0	7.7	4.6	21.0	9.0	5.5
	Leave	12.5	27.4	13.0	5.0	22.0	8.9	4.1
	Salvage	0.5	14.9	5.6	5.4	15.2	3.8	2.5

¹ Measurements obtained from CWD > 8cm in diameter at transect point of intersect (see methodology for details).

² Mean number of CWD per 22.6m transect within a 0.04ha plot (n = 42).

³ Mean diameter of CWD at point of transect intersect.

⁴ Mean rot class ranking of CWD:

1 = Log whole and undecayed, bark, branches and twigs present and intact, wood hard.

2 = Log sound, wood hard, twigs mostly lacking, <50% bark missing;

3 = Some branches remaining, >50% bark missing, wood soft in places

4 = Little to no bark remaining, no branches, fungi present, wood soft with small crevices, some pieces lost;

5 = Large wood fragments lost, outline of trunk deformed, log covered in moss, vascular plants beginning to colonise

6 = Wood mostly decayed, completely moss covered and covered in plants;

7 = Humification nearly 100%, hard to define as a log, no evidence of hard wood.

Appendix 10: Total number of species detections during "Interval 2" of the Black-capped Chickadee mobbing playbacks that ranked 1 or ≥ 2 and the number of observations that resulted in additional information gained from "Interval 3".

Species	Controls			Leaves			Salvages		
	Rank 1 (Int. 2)	Rank ≥ 2 (Int. 2)	Add. Obs. (Int. 3)	Rank 1 (Int. 2)	Rank ≥ 2 (Int. 2)	Add. Obs. (Int. 3)	Rank 1 (Int. 2)	Rank ≥ 2 (Int. 2)	Add. Obs. (Int. 3)
AMKE	0	0	0	0	0	1	0	0	0
AMRE	1	2	0	2	0	0	0	0	0
AMRO	1	0	1	0	1	1	2	0	0
BBWO	0	0	0	0	0	0	1	0	0
BCCH	3	27	0	6	17	0	4	2	0
BGNW	1	0	1	0	0	0	0	0	0
BOCH	5	10	0	1	0	0	0	0	0
BHVI	5	1	1	1	7	1	0	1	1
BPWA	0	0	0	2	0	0	0	0	0
BRBL	0	0	0	0	0	0	1	0	0
BRCR	0	0	1	4	3	2	0	0	0
CEWX	0	0	0	0	0	2	1	0	3
CHSP	2	0	2	4	2	1	6	0	0
COSN	0	0	0	0	0	0	0	0	2
COYE	0	0	0	4	0	1	0	0	0
DEJU	3	0	1	3	3	0	3	1	2
DOWO	2	0	0	3	0	0	0	0	0
EMPI*	0	0	0	2	0	0	0	0	0
GCKI	9	4	6	4	1	1	0	0	0
GRJA	0	1	2	1	4	3	0	0	0
HAWO	0	0	2	4	0	2	0	0	0
HOWR	0	0	0	2	0	0	0	0	0
LCSP	0	0	0	1	0	0	5	1	3
LEFL	0	0	1	2	1	1	0	0	0
LISP	0	1	0	10	6	2	20	4	4
MOWA	6	0	4	9	2	4	1	0	0
NOFL	0	0	0	4	2	2	0	0	0
OSFL	0	0	0	0	0	0	0	0	1
OVEN	11	0	2	0	0	1	0	0	0
PHVI	3	2	0	0	0	0	0	0	0
PISI	3	1	4	2	8	8	2	6	3
RBGR	1	0	0	0	1	1	0	0	0
RBNU	14	11	7	10	16	2	1	1	0
RCKI	0	0	0	7	2	0	0	0	0
REVI	4	1	1	0	0	0	0	0	0
RWBL	0	0	1	1	0	2	0	0	0
SOSA	0	0	0	0	0	0	0	0	1
SWTH	12	4	0	4	0	2	0	0	0
TEWA	15	4	1	5	2	1	0	0	0
TRSW	0	0	0	0	0	0	1	0	0
TTWO	0	0	0	2	2	0	0	0	0
VISP*	0	2	0	0	0	0	0	0	0
WAVI	10	1	1	3	3	1	0	0	0
WBNU	0	1	0	0	1	0	0	0	0
WETA	0	0	0	0	1	0	0	0	0
WTSP	6	1	9	9	2	6	3	3	0
WWPE	0	0	0	7	0	1	0	0	1
YBSA	4	1	0	0	0	1	1	1	0
YRWA	25	9	0	12	5	1	2	0	0
YWAR	1	0	0	0	0	0	0	0	0

Appendix 11: Number of cavity nests, separated into nesting guilds, located throughout the Chip Lake study area from the 1st (1999) to 3rd (2001) year post-fire. Numbers in brackets are nests located outside of 10 ha study sites but in similar treatment habitat. C = Controls (unburned/unharvested), L = Leaves (burned/unharvested), S = Salvages (burned/harvested).

Guild ¹	Species ²	1999			2000			2001		
		C	L	S	C	L	S	C	L	S
PCN	BBWO	----	1	----	----	(2)	----	----	(1)	----
	TTWO	----	2 (3)	1	----	3 (6)	1	(1)	(3)	----
	HAWO	(1)	2	1	1	3 (3)	(1)	1	2 (1)	----
	DOWO	----	----	----	(1)	1 (1)	----	----	----	----
	PIWO	----	----	----	(1)	----	----	----	----	----
	NOFL	(3)	2	----	1	4	6 (7)	----	2 (2)	2 (3)
	YBSA	2 (1)	3	----	6 (7)	3 (2)	----	2 (4)	2 (2)	----
WCN	BCCH	----	----	----	----	1	----	1	----	----
	BOCH	----	----	----	----	1	----	----	----	----
	RBNU	----	----	----	1 (7)	3 (1)	----	4 (3)	9 (1)	(1)
SCN	AMKE	----	----	----	----	3	1	----	4	1
	BRCR	----	----	----	----	2	----	1	2	----
	DUCK Spp.	----	----	----	1	(1)	----	----	2	----
	EUST	----	----	----	----	(1)	----	----	(1)	----
	HOWR	----	6	3 (2)	----	17 (2)	2 (1)	(1)	4	1 (3)
	MOBL	----	----	----	----	----	----	----	(1)	1 (1)
	NHOW	----	----	----	----	1 (1)	----	----	(1)	----
	TRSW	----	----	(1)	----	1 (1)	1(4)	----	1 (1)	(2)
	RESQ	----	----	----	1	----	----	----	(1)	(1)

¹ PCN = Primary cavity-nesters, WCN = Weak cavity-nesters, SCN = Secondary cavity-nesters (see Chapter 4).

² See Appendix 2 for species codes.